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**Biogeochemistry of Mercury in a
River-Reservoir System: Impact of
an Inactive Chloralkali Plant on
the Holston River-Cherokee
Reservoir, Virginia and Tennessee**

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ENVIRONMENTAL SCIENCES DIVISION
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BIOGEOCHEMISTRY OF MERCURY IN A RIVER-RESERVOIR SYSTEM:
IMPACT OF AN INACTIVE CHLORALKALI PLANT ON THE HOLSTON
RIVER - CHEROKEE RESERVOIR, VIRGINIA AND TENNESSEE¹

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ENVIRONMENTAL SCIENCES DIVISION
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ABSTRACT

HILDEBRAND S. G., S. E. LINDBERG, R. R. TURNER, J. W. HUCKABEE, R. H. STRAND, J. R. LUND, and A. W. ANDREN. 1980. Biogeochemistry of mercury in a river-reservoir system: Impact of an inactive chloralkali plant on the Holston River - Cherokee Reservoir, Virginia and Tennessee. ORNL/TM-6141. Oak Ridge National Laboratory, Oak Ridge, Tennessee. 160 pp.

Elevated mercury concentrations in fish species from the North Fork of the Holston River were observed in the early 1970's. The source of the mercury was a chloralkali plant which had ceased operation in 1972. Mercury continues to be released to the river from two large (~ 40 -ha) waste disposal ponds at the plant site. This report presents results of our study of the emission of mercury to the environment from the abandoned waste ponds and of the distribution of mercury in water, sediment, and biota of the Holston River-Cherokee Reservoir System in Virginia and eastern Tennessee.

The major component of the waste pond material is calcium carbonate. Total mercury concentration in the solid waste material ranged from 4.4 to 350 $\mu\text{g/g}$. The total mercury concentration in leachates from the waste ponds ranged from 11 to 230 $\mu\text{g/liter}$. Annual mercury emission rates from the waste ponds have been estimated to be ~ 36 kg/year to the atmosphere and ~ 39 kg/year in leachates from the ponds. The atmospheric emission of mercury from the ponds consisted primarily of mercury vapor, was related to surface temperature, and during warm months might result in air concentrations approaching the Environmental Protection Agency guideline of 1000 ng/m^3 in ambient air near the ponds. The mercury concentration in air appeared to decrease to background levels within ~ 1.5 km of the waste ponds.

The concentrations of mercury in water and in suspended sediments of the North Fork Holston River downstream from the plant site were typically 20 times higher than immediately upstream. The silt-clay fractions of bottom sediment immediately downstream of the plant contained up to 200 times as much mercury as immediately upstream. Mercury concentrations in water, suspended matter, and bottom sediment decreased rapidly to a point ~ 40 km below the plant, and then they varied only slightly. Dilution accounted for approximately 50% of the decrease with downstream distance. Mercury concentrations in sediment cores from Cherokee Reservoir (~ 200 km downstream from the plant) indicated that mercury input to this reservoir was highest during early plant operation.

Concentrations of mercury in fish species and benthic invertebrates in the North Fork Holston River were lowest above the plant, peaked immediately downstream from the plant in the range of 1 to 2 $\mu\text{g/g}$, and then decreased further downstream. Mercury concentration in fish species 133 km below the plant was still above the U.S. Food and Drug Administration "Action Level" of 0.5 $\mu\text{g/g}$ in effect when this study was conducted. The majority of total mercury in fish muscle in the Holston-Cherokee system was in the methylated form, while $\sim 50\%$ of the mercury in invertebrates was methylmercury. Mercury concentration in fish was related to fish weight in some but not all collections. A multiple regression analysis indicated that 80 to 90% of the variance in mercury concentration in fish species from the North Fork could be explained by concentration of mercury in water, sediments, and benthic invertebrates. Concentration of mercury in fish

and invertebrates was well correlated with sediment mercury concentration. The potential implications of this relationship for mercury monitoring programs are discussed.

Fish species analyzed for mercury in Cherokee Reservoir ~ 250 km below the plant generally contained mercury in muscle at levels < 0.5 $\mu\text{g/g}$. Fish species that did exceed 0.5 $\mu\text{g/g}$ were generally from the upstream end of the reservoir.

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1. INTRODUCTION

This research on the biogeochemistry of mercury in the Holston River - Cherokee Reservoir system was initiated in 1974 as part of the Ecology and Analysis of Trace Contaminants Program at Oak Ridge National Laboratory. In the early 1970's, elevated levels of Hg in fish in the North Fork Holston River were observed by the State of Virginia and the Tennessee Valley Authority. The suspected main source of mercury contamination was a chloralkali plant located in Saltville, Virginia, on the North Fork of the Holston River ~ 200 km upstream from Cherokee Reservoir. The chloralkali plant was decommissioned in 1972 due primarily to the high cost of pollution abatement.

The major objectives of the study reported here were to quantify the source of mercury inputs to the Holston River - Cherokee Reservoir system from the chloralkali plant and to determine the downstream distribution and transport of mercury in biotic and abiotic compartments of the aquatic system. As the study progressed, quantification of possible atmospheric emissions of mercury from waste ponds at the plant site was added to the study objectives.

Chapter 2 of this report presents results of research on the chemistry and emission of mercury from abandoned waste ponds at the plant site. Chapter 3 addresses downstream distribution and transport of mercury in water and sediments in the Holston - Cherokee System. Chapter 4 reports results of research on the accumulation of mercury by fish and invertebrates. A summary of major observations as a result of the overall study is presented in Chapter 5. Chapters 2 to 4 are

self-contained (to the extent practical) and include appropriate introductory material and methods relevant to each chapter.

This research was funded by the National Science Foundation's Research Applied to National Needs (RANN) Program. Major results of this study have been published previously (Hildebrand et al. 1975, Lindberg and Turner 1977, Turner and Lindberg 1978, Hildebrand et al. 1980). Our purpose in preparing this document is to present all original data and results of our research in one document with detailed discussion so that a complete record will be available. The information in this document, coupled with the ongoing research efforts of the State of Virginia, State of Tennessee, and the Tennessee Valley Authority, should provide useful criteria to judge the success or failure of ameliorative strategies implemented at the old chloralkali plant site and waste disposal areas.

2. CHEMISTRY AND EMISSION OF MERCURY FROM CHLORALKALI SOLID WASTE PONDS

Steven E. Lindberg, Ralph R. Turner, and John R. Lund

INTRODUCTION

The role of the chloralkali industry in global and regional mercury cycling has often been discussed, with the general conclusion that its contribution may be of limited importance globally, but of particular concern to local and regional water and air quality. Although limited studies have been published on losses of Hg from production processes of active chlorine plants, none have considered the problem of atmospheric and aquatic emissions from waste products which have been ponded or otherwise stored near active or inactive plants. This chapter describes the chemistry of Hg in the aqueous and solid phases of wastes stored near an inactive plant. A major thrust of this research was to determine the mechanisms and rates of Hg emissions from these waste deposits.

Study Site

The former site of the chemical manufacturing complex, which included a chloralkali (mercury cell process) and ammonia soda plant, is in southwestern Virginia (Fig. 1) on the banks of the North Fork of the Holston River (NFHR). The NFHR below the plant site has been closed to fishing, except on a catch-and-release basis, since 1970 because fish taken from this portion of the river continued to exhibit mercury levels which exceeded the contemporary FDA limit of 0.5 $\mu\text{g/g}$ in

Fig. 1. Location of the inactive chloralkali plant on the North Fork Holston River showing location of the waste ponds (RK0 represents the river kilometer zero point for reference; the mass balance stations are discussed later in this section). The pond on the right is designated Pond 1 for future reference.

edible flesh. Since it has been proposed (Bailey 1974, Hildebrand et al. 1975, Toole and Ruane 1976, Lindberg and Turner 1977) that this problem persists because of continued significant losses of mercury from residual wastes stored at the plant site, some description and recent history of the plant site is necessary. Two large unlined settling ponds were used for clarification of waste slurries from the ammonia soda, caustic soda, and chlorine processes, with the ammonia soda plant contributing by far the largest amount of solid waste. The chlorine plant, which was built in 1952 with an annual design capacity of 81,000 metric tons, (Congressional Record 1970) was closed in June 1972 largely due to the high cost of retrofitting for adequate pollution abatement. Neither production nor operational information was obtainable from the parent company. The chlorine plant was partially dismantled in the spring of 1973 with some unsalvageable components, including the intact mercury cells and contaminated earth from the cell room site, that were buried in settling pond number 2. The balance of the plant was finally razed in the spring of 1976.

The settling ponds which cover 44 ha are situated immediately adjacent to the NFHR behind steep dikes composed of boiler ash and CaCO_3 waste (Fig. 2). The total volume of solid waste in the settling ponds is estimated by us to be $\sim 6 \times 10^6 \text{ m}^3$ based on the surface area of each pond and their reported geometry and thickness (Saltville Planning Commission 1973). The location of the ponds perched above the NFHR and on the flank of an adjacent ridge favors leaching of soluble waste components by direct rainfall and by runoff from the ridge. The ponds are drained from the surface and at

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Fig. 2. Aerial view of the waste disposal ponds perched above the NFHR behind man-made dikes.

intermediate depths by stop log structures while clay, iron, and plastic pipe provide drainage through the bottom of the dikes. Discharge of pond leachate through these structures and from many intermittent seeps from the face of the dike flows directly into the river.

Methods and Materials

Sample Collection

The results reported here are based on analyses of water, settling pond solids, and leachates collected between February 1975 and January 1976. All unfiltered water and leachate samples were collected directly in glass, pre-baked ($@ 500^{\circ}\text{C}$) and pre-acidified ($\text{HNO}_3 + \text{K}_2\text{Cr}_2\text{O}_7$), volumetric flasks to minimize sorption and contamination. Where filtering was required for separation of suspended matter, an all-teflon/nylon pressure (N_2 gas) filtration apparatus fitted with a 10.2-cm-diameter, 0.45-micron membrane filter (Gelman Metrical) was employed for rapid (1 liter in 2-3 min) field filtration, the filtrate passing directly into a volumetric flask prepared as above. All solid samples were refrigerated (4°C) and returned to the laboratory where they were homogenized and dried ($< 50^{\circ}\text{C}$) prior to analysis for mercury. Ambient air samples for particulate and vapor Hg were collected, using standard air filter and activated charcoal adsorption techniques respectively, and analyzed as described below.

Measurement of Hg Vapor Emission

Measurement of the emission rates of Hg vapor from solid waste was performed using a flow-through chamber system originally described in Lindberg and Turner (1977). The system utilizes the affinity of activated charcoal for vapor phase Hg to measure the net flux of Hg from a known soil surface area. The ability of activated charcoal to remove gaseous Hg is well known (Sergeant et al. 1957), and similar techniques have been applied to monitor Hg emissions and ambient air Hg concentrations near known sources (Moffitt and Kupel 1971, Rathje et al. 1974, Lindberg and Turner 1977).

Figure 3 describes the apparatus used in these experiments. The sampling train consists of an outside air intake tube and flow meter [(a) and (b)] through which ambient air is carried to the 9-liter glass dome (e) after being scrubbed of its initial Hg content by a set of 2 activated charcoal sampling tubes and an air filter (c). Air flows turbulently across the soil surface (d) and passes out through an air filter and a series of two activated charcoal tubes (f) maintained slightly above ambient temperature to prevent condensation of water vapor onto the charcoal. Air is drawn through the system at a constant 1-liter/min rate using an air-pump/critical-flow orifice system (h).

In practice, the entire system including waste to be tested was maintained at a constant temperature for 24 h prior to initiation of the measurements. At this time the dome was placed over the waste, sealed, and ambient air drawn through the system from inlet (a). After flushing for 10 min, the two activated charcoal sampling tubes were placed at the dome outlet and the experiment initiated. The sampling

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Fig. 3. Description of the apparatus designed for experimental determination of Hg volatilization from waste material.

^aOutside air intake, at atmospheric pressure.

^bCalibrated rotameter.

^cPair of activated charcoal Hg° absorption tubes in series to remove Hg° from incoming air stream.

^dSample container with surface thermometer.

^eSealed glass air flow chamber, internal temperature controlled by the use of heat lamps or mechanical refrigeration.

^fPair of activated charcoal Hg° absorption tubes in series for collection of Hg° evolved from waste surface (tubes kept above ambient air temperature to prevent condensation of water vapor onto charcoal).

^gCondensation traps.

^hAir pump and critical flow orifice to maintain flow rate at constant rate of 1-liter/min.

times ranged from 8 to 24 h during which time the surface waste temperature within the dome fluctuated $\pm 2^{\circ}\text{C}$ of the stated value and the flow rate fluctuated ± 0.2 liter/min of the stated 1 liter/min. All reported temperatures and flow rates are, thus, mean values. Following completion of an experiment, the two outlet charcoal tubes were analyzed separately for total Hg.

The efficiency of the activated-charcoal traps to remove volatile Hg was determined by placing the tubes in line with a standard flameless atomic absorption spectrometer system (described in the following section) and injecting known quantities of elemental Hg vapor into the system. With this method it was possible to investigate the effect of flow rate, sampling time, air stream temperature, and relative humidity on efficiency and to calculate capacity of the tubes. Briefly, these results indicated the following: (1) regardless of sampling conditions the first tube in series contained $96 \pm 4\%$ of the total collected by both tubes; (2) overall collection efficiencies for two new tubes in series at 1-liter/min flow rates and maintained slightly above ambient temperature to prevent water vapor condensation on the charcoal ranged from 98 to 100% for atmospheres containing from 2 to $700 \mu\text{g}/\text{m}^3$ Hg⁰; (3) the capacity of the activated charcoal for Hg is $\sim 1\%$ of the charcoal weight, where each tube generally contains ~ 1.5 g of charcoal; (4) collection efficiency decreases as flow rate increases and also as water vapor condenses on the charcoal.

Mercury Determinations

Laboratory analyses for Hg followed the procedure given by Hatch and Ott (1968) with modifications including the digestion procedure for both solid and aqueous samples as described in Feldman (1974). Solid samples were homogenized by hand and replicate portions placed into prebaked and acid-washed borosilicate glass containers where they were wet ashed with a HNO_3 , HClO_4 , and $\text{K}_2\text{Cr}_2\text{O}_7$ solution using jacketed condensers as described in Feldman (1974). Aqueous samples were done in similar glassware and digested as above following filtration through 0.4- μm Nuclepore membrane filters. Plant material was wet ashed and analyzed in toto. The analytical system included a 1.0-m optical path length and afforded a limit of detection of 0.5 ng Hg. Thus mercury could be reliably detected in water at the 0.025- $\mu\text{g/liter}$ level without preconcentration. Sample concentration by evaporation using jacketed air condensers (Feldman 1974) permitted reliable detection of mercury in water down to 0.001 $\mu\text{g/liter}$. Accuracy as judged by digestion and analysis of NBS standard orchard leaves was $\sim 94\%$, while precision based on replicate portions of solid material carried separately through the entire digestion and analytical technique ranged from ± 1 to $\pm 10\%$.

The standard method for analysis of activated charcoal for Hg, which involved an HCl-HNO_3 leach, was found to result in erratic recoveries. The problem was related to formation of stable iodo-mercury complexes in the charcoal/acid solution. To reduce this problem the following technique was developed: activated charcoal from one sampling tube was placed in a prebaked volumetric, 10 ml of H_2SO_4

was added, and the mixture heated in a reflux condenser as described above until the I_2 vapors disappeared (generally 1-2 h). This solution was then cooled and carried through the normal digestion procedure as described above. Using this method, recoveries were determined with both spiked charcoal/acid solutions and charcoal tubes of known Hg contents, resulting in recoveries ranging from 70 to 102%.

Supplementary analyses of water for major cations, anions, and suspended solids followed APHA (APHA 1971) or equivalent procedures. X-ray diffraction (XRD) studies of solids employed a Debye-Scherrer powder camera.

Results and Discussion

Solid and Liquid Waste Chemistry

The major constituent composition of the solid waste and aqueous leachate from the settling ponds is summarized in Table 1. Because the bulk of the waste found in the slurry ponds originated from the soda production plant, the occurrence of calcium as the major component of the waste is not surprising since carbonate precipitation was the major liquid waste purification process (C. Arnold, personal communication). X-ray diffraction analysis of the waste confirmed that the predominant crystalline phase (40-80% of total mass) was calcium carbonate (calcite). The presence of the other elements is typical of wastes produced in the manufacture of sodium bicarbonate, caustic, and chlorine. X-ray diffraction data also indicated that some of these of these secondary elements in the waste are bound in

$Ca_{6-x}Mg_xAl_2(SO_4)_3(OH)_{12} \cdot 25H_2O$ which appeared to account for 10 to 40%

Table 1. Major chemical composition of muck waste and natural leachate from the chloralkali ponds

	Solid waste (% composition)	Waste pond leachate (mg/liter)
Na	0.2 - 0.3	5,900
Ca	25 - 33	2,300
K	1 - 3	20
Mg	2 - 3	1
Cl	4 - 7	12,000
SO ₄ ⁼	1 - 10	260
OH	ND ^a	170 ^b

^aND = not determined.

^aCalculated from leachate pH = 12.

of the waste. The remainder of the material likely includes a number of double or triple compounds of Na, Ca, Mg, Al, Fe, and Si present as oxides, hydroxides, carbonates, chlorides, sulfates, or silicates (Hou 1942).

The composition of the natural leachate from the settling ponds indicates it to be a caustic brine composed primarily of chlorides and hydroxides of sodium and calcium. Charge balance calculations indicate a slight excess of total cations (372 meq/liter) over anions (352 meq/liter). Assuming the solution to be saturated with CaCO_3 would account for only 1.6 meq/liter of $\text{CO}_3^{=}$. Other possible contributing anions such as $\text{NO}_3^- + \text{PO}_4^{=}$ = contributed < 1 meq/liter to the solution. Thus, given the sample dilution required prior to analysis for the major constituents, a combined error of $\pm 6\%$ in the charge balance is not unreasonable.

Total mercury content of the solid waste collected in cores from the upper 2 m of one of the settling ponds varied from 4.4 to 350 $\mu\text{g/g}$, with an average of 92 $\mu\text{g/g}$ (see Table 2). The limited core data for the settling ponds generally indicated higher mercury concentration near the surface of the waste than at depth. Although this observation is not critical to the following discussion, it could reflect the increased efficiency with which this facility was able to remove Hg from its waste process streams prior to river discharge during its final years of operation.

The complex chemistry of the solid waste, coupled with the occurrence of Hg as a relatively minor component, precluded a straightforward qualitative analysis of the form of Hg present in the

Table 2. Vertical distribution of mercury in a surface core from one chloralkali waste pond

Sampling interval (cm)	Total Hg ($\mu\text{g/g}$)
0 - 2	179
2 - 5	97
5 - 14	87
14 - 17	193
17 - 20	161
20 - 24	340
24 - 30	200
30 - 33	189
33 - 36	240
36 - 38	290
100 - 110 ^a	76
110 - 120	106
120 - 130	73
130 - 140	63
140 - 150	68
150 - 160	105
160 - 170	17
170 - 180	9
180 - 190	4
190 - 200	28
200 - 210	14

^aIntervals below 100 cm must be considered approximate due to the possibility of compaction during coring.

solid phase. A grey-black precipitate which was collected from the side of an outfall pipe, containing 0.4% Hg by weight, was subjected to density gradient separation and X-ray diffraction (XRD). Even after further concentration by density gradient separation, no crystalline Hg compounds could be identified. Chemical thermodynamic considerations suggest Hg could occur to some extent in the solid phase as basic carbonates or chlorides such as $\text{HgCO}_3 \cdot 2\text{HgO}$ or $\text{HgCl}_2 \cdot 3\text{HgO}$. However, laboratory studies of the emission of mercury vapor (Hg^0) from the waste as a function of temperature suggested that mercury exists in the waste to some extent as the elemental species, probably dispersed in the waste in a finely divided form (see discussion in this chapter). The volatility of this species indicates that there could have been some loss of Hg during our drying of the waste at 50°C . Thus, the reported solid waste Hg concentrations may represent minimum values.

Total mercury in leachates collected from the ponds varied from 11 to 230 $\mu\text{g/liter}$ and averaged 68 $\mu\text{g/liter}$. The data presented in detail in Table 3 indicate that (1) only one of the ponds (Pond 1) appears to be a major source of mercury and (2) the highest mercury concentrations are associated with the pipe outfall from Pond 1 which receives water from both the pond surface and subsurface seepage. The aquatic flux of mercury from the settling ponds into the adjacent river via seepages and overflow structures (pipe outfalls) is discussed in detail in a following section. Briefly, the net flux of mercury from the disposal site via aquatic pathways, as measured by a mass balance approach, varied from 26 to 350 g/d, the higher fluxes being associated with stormflow conditions. By comparison, on one day while the plant was

Table 3. Mercury concentrations and wastewater flow rates of major outfall pipes and accessible seepages in the chloralkali waste ponds

Location	Date	Flow (liter/min)	Total Hg ($\mu\text{g/liter}$)	Total Hg (flux g/day)
<u>Pond 1</u>				
Pipe outfall	5/20/75	-	84	-
Pipe outfall	7/31/75	53	103	7.9
Pipe outfall	1/15/76	144	230	47.7
Pipe outfall	1/16/76	156	220	49.4
Seepage 1+2 ^a	7/31/75	1.7	43	0.1
Seepage 3+4 ^a	7/31/75	2.1	37	0.1
Seepage 5	7/31/75	6.4	51	0.5
Seepage 6+7 ^a	7/31/75	3.5	42	2.2
Seepage 8+9 ^a	7/31/75	9.4	41	0.6
Seepage 10	7/31/75	6.6	47	0.4
Seepage 11	7/31/75	5.9	45	0.4
Seepage 12	7/31/75	8.6	16	0.2
Seepage 13	7/31/75	5.6	13	0.1
Seepage 14	7/31/75	8.4	11	0.1
Seepage 15	7/31/75	33.6	29	1.4
<u>Pond 2</u>				
Pipe outfall	5/20/75	-	0.09	-
Pipe outfall	1/15/76	13	0.06	0.001
Seepage 1	1/15/76	12	0.07	0.001
Seepage 2	1/15/76	0.8	1.18	0.001
Seepage 3	1/15/76	6.9	0.42	0.004
Seepage 4	1/15/76	44	7.4	0.5
Combined seepage	5/20/75	-	3.2	-
Combined seepage	1/15/75	0.2	2.1	0.001

^aRepresents proportional composite of two adjacent seeps.

still in operation (8/12/70), the flux of mercury into the river was reported to be 260 g/d. For the period October 1973 to September 1974, the estimated annual aquatic flux of mercury from the disposal site was ~ 40 kg. Significantly, a substantial fraction (25 - 50%) of the total aquatic flux of mercury from the settling ponds was contributed by flow from the single pipe described above.

According to general thermodynamic equilibria considerations (Hem 1970, Hahne and Kroontje 1973) the high pH and chloride content of the leachate strongly favor existence of dissolved mercury in the leachate as hydroxy- and chloro- complexes as well as dissolved Hg^0 . The outfall sample collected from Pond 1 on 1/16/76 was analyzed for the major constituents listed in Table 2, as well as for Hg. These values were utilized to compute possible equilibrium species of Hg using MINEQL, a computer program for the calculation of chemical equilibrium composition of aqueous systems (Westall et al. 1976). These calculations supported the probable existence of the above complexes, predicting the following: (1) the dissolution of Hg^0 present in the solid waste and in contact with the leachate solution; and (2) the existence, at equilibrium, of $\sim 70\%$ of the aqueous Hg as $\text{HgCl}_4^{=}$, $\sim 20\%$ as HgCl_3^- , and the remainder as HgCl_2 and Hg(OH)_2 .

Although leachate samples were not filtered, in no case did visual inspection indicate the presence of a particulate fraction. Thus we have assumed that Hg is leaching from the immediate pond area in a dissolved phase. However, upon mixing with the river, small plumes of white floc or precipitate have been seen to form, generally within 1 m of the outfall pipe. Given the possible degree of carbonate saturation

of the leachate, it is not unreasonable to expect precipitation of CaCO_3 onto various nucleation sites on the suspended matter in the river. Such a reaction would be expected to remove Hg^{++} ions from solution to some extent, although the ability of calcite to sorb Hg^{++} has apparently not been reported in the literature.

To further study the behavior of soluble Hg in the solid waste, selected samples were batch-extracted with rainwater. A 40-cm core of the solid waste, collected when the settling pond contained no standing water, was split in the laboratory and a surface (0-2 cm) and subsurface (36-38 cm) sample removed. The moist waste was oven-dried at 50°C , thus incorporating any pore-water-dissolved Hg into the solid fraction, and homogenized by hand-grinding with a mortar and pestle. Subsamples of 2.0 g from each split were then swirled for 96 h on a rotary shaker at 25°C with 100 ml of rainwater (initial pH = 3.8, dissolved Hg = $0.13 \mu\text{g/liter}$). For comparison, a sample of intact core (0-10 cm) was centrifuged at 3000 rpm for 30 min to isolate the interstitial water fraction for similar analysis. Both sample types were filtered as described previously prior to analyses.

The results of these experiments (Table 4) illustrate two important points: (1) solid and extractable Hg concentrations are highly variable even on a small spatial scale, and (2) very little of the Hg present in the waste is in a readily leachable state under these experimental conditions. The samples from the surface of the core exhibited lower solubility of both whole waste (1%) and Hg (0.1 - 0.9%) than subsurface samples (4% of solid and 2% of Hg). This trend may simply reflect the inclusion of salts dissolved in the relatively

Table 4. Summary of laboratory mercury-leaching experiment using chloralkali solid wastes

Sample depth (cm)	Total Hg in solid (μg/g)	Dissolved Hg (μg/liter)	pH	Batch leachate			Fraction of total waste soluble (%)	Fraction of total Hg soluble (%) ^a
				Cl	Na	Ca		
				(μg/liter)				
0 - 2 (split #1)	230	6.4	8.6	45	-	-	1.2	0.14
0 - 2 (split #2)	179	34	7.9	22	17	10	-	0.95
36 - 38 (split #1)	160	54	10.2	122	-	-	4.4	1.8
36 - 38 (split #2)	290	133	8.4	99	31	140	-	2.3
0 - 10 (interstitial water)	55 ^b	1160	12.4	2184	1280	390	-	1.6

^aIncluding mercury dissolved in interstitial water of waste.^bSolid waste sample was 75% moisture by weight.

higher moisture content of subsurface samples rather than actual differences in waste and Hg solubility.

The interstitial water sample provides an interesting comparison as it represents a much higher solid/solution ratio (~ 0.3 g/ml) than the batch extraction experiments (0.02 g/ml). However, on a relative scale, approximately the same proportion of the total Hg present was dissolved in the pore water as was extracted by the batch experiments. This suggests that for these surface waste samples (upper 1 m) a small, relatively constant, fraction of the total Hg is readily soluble, existing primarily in the interstitial moisture of the waste.

Comparison of the concentrations in the batch extractions with those measured in the actual waste pond seeps affords some insight into the possible source of the seep solution within the ponds. The concentration of Hg in the laboratory rainwater extractions (6-130 $\mu\text{g/liter}$) were similar to those observed in the pond seepage (11-230 $\mu\text{g/liter}$). The relative solubility of the waste-associated Hg in rainwater is of interest because the subsurface moisture pool, recharged from above by rainwater percolation and depleted from below by drainage, is thought to be the primary source of the leachate entering the river. Considering the field situation, continuous column extractions of the waste may have been a better analog than batch extractions.

Emission of Hg From the Waste Site - Atmospheric Sources

The volatility of metallic Hg (Hg^0) and many of its compounds suggests its presence in large-scale waste deposits such as these to be

a potential threat to local air quality. Waste material collected from the surface of the waste ponds was used in laboratory experiments to determine the emission of Hg^0 from a known surface area at controlled temperatures. Approximately 4 kg of waste material (with a total Hg concentration of 155 $\mu\text{g/g}$) was slurried into a 19-liter glass chamber and allowed to dry to a moisture level similar to that observed for the surface of the actual waste ponds (approximately 75% water by weight). Mercury-free air was pumped through the chamber over the waste surface at 1 liter/min and the filtered output air sampled for Hg^0 . A schematic drawing of the experimental apparatus was described earlier in Fig. 3. The emission of elemental Hg vapor from the waste material was clearly related to surface temperature as expected (Fig. 4). The nature of the relationship is similar to that between temperature and the saturation concentration of Hg^0 in air (Wallace et al. 1971), suggesting that the mercury occurs in the waste material to a large extent in the elemental state. Comparative data on Hg emission rates from natural surfaces are scarce, although some authors have reported observing release from soils experimentally enriched with inorganic mercury (Gilmour and Miller 1973, Johnson and Brahman 1974, Williston 1968). Natural degassing rates of Hg on regional or global scales have been estimated based on input/output calculations as ranging from 0.02 to 0.03 $\mu\text{g/m}^2\text{h}$ (Weiss et al. 1971, Kothny 1973), while measured rates for mineralized soils are as high as 1.7 $\mu\text{g/m}^2\text{h}$ (McCarthy et al. 1969). We have recently measured the loss of Hg from contaminated surface soils (total Hg = 99 $\mu\text{g/g}$) collected within 1 km of a large active Hg mine, and found emission rates at 33°C to be in the range of

Fig. 4. Relationship between emission of vapor phase elemental mercury and surface temperature of waste material. The numerical relationship derived from a nonlinear least-squares fit is $Y = a + b \exp(cX)$, where Y = emission rate in $\mu\text{g/m}^2\text{-h}$, X = surface temperature in $^{\circ}\text{C}$, $a = 6.40$ (standard error = 1.59), $b = 0.310$ (standard error = 0.100), $c = 0.190$ (standard error = 0.00982).

0.5 to 0.7 $\mu\text{g}/\text{m}^2\text{h}$ (Lindberg et al. 1979). These values are approximately two orders of magnitude lower than those we have measured under comparable conditions for the waste material (120-170 $\mu\text{g}/\text{m}^2\text{h}$). The difference is likely related to the form of Hg present in the two samples; the mine site soil contains primarily HgS and HgO.

Considering the waste deposits as an area source, we have applied a simple Gaussian diffusion model (Gifford 1968) to estimate 24-h mean concentrations of Hg vapor in the ambient air downwind of the source (R. J. Raridon, personal communication). The emission rates measured under controlled conditions (Fig. 4) were used in the model; mean wind speed directly over the pond area was assumed to be 1 m/sec; and the background air Hg concentration was taken as 10 ng/m^3 . For comparative purposes, air sampling stations were established near the ponds on two occasions. These were equipped to measure Hg^0 in air, Hg on air particulates, wind speed and direction, air temperature, and bulk deposition (Fig. 5). Table 5 presents a comparison of measured and predicted air concentrations of mercury vapor. Given the simplicity of the modeling approach, the close agreement between the measured and predicted 24-h mean air concentrations under similar conditions can be taken as an indication that the entire surface area of the waste pond was emitting Hg vapor at a rate comparable to that measured in controlled experiments.

The levels of Hg^0 in air measured near the pond during periods of low air temperature (Table 5) were slightly elevated above those reported for large U.S. cities (Johnson and Brahman 1974, Van Horn 1975). The fact that these concentrations decreased to rural

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Fig. 5. Location of the atmospheric chemistry sampling station at the edge of one waste pond. Equipment utilized included aerosol and gas samplers for particulate Hg and Hg⁰, wind vane and anemometer, and bulk deposition collectors.

Table 5. Daily mean concentrations of mercury vapor (Hg^0) in air in vicinity of the waste ponds under various meteorological conditions

Distance from center of waste ponds (km)	Percent of time downwind ^a	Mean air temperature	Concentration of mercury vapor (Hg^0) in air	
			Measured (ng/m^3)	Predicted (ng/m^3)
0.4	78	6.3°C	90	65
0.5	99	6.3	64 ± 6 ^b	72
1.5	67	6.3	18	26
1.9	21	6.3	12	15
0.5	96	29.4	991 ± 2 ^b	778

^aExpressed as a percent of the total sampling period (24 h) during which the sample location was downwind of the waste pond.

^b₁₁ = 3.

background levels (Kothny 1973) within 1.5 km of the waste disposal area is attributable to the ability of a well-mixed atmosphere to rapidly dilute gaseous pollutants. However, during periods of higher air temperature, the concentration of Hg measured near the pond area approached the U.S. Environmental Protection Agency guideline level for total Hg in ambient air (1000 ng/m^3) (Federal Register 1973). This compares with levels of 30 to 1600 ng/m^3 for air over natural Hg deposits and 10 to $28,000 \text{ ng/m}^3$ for natural degassing and geothermal emissions (Kothny 1973). The concentration of particulate Hg during this sampling period was surprisingly low (0.25 to 0.36 ng/m^3) considering the area in question, and reflected the stability of the waste pond surface to dust resuspension. The indication that essentially 100% of the Hg in air is in vapor form agrees with recently reported results for ambient air and industrial power plant emissions (Lindberg 1980).

Given the elevated levels of Hg^0 measured near the pond during the high temperature period, the diffusion model (Gifford 1968) can also be used to determine approximate isopleths of Hg^0 concentration in residential areas near the pond during a typical hot weather period. The results of this calculation are presented in Fig. 6 which illustrates mean hourly Hg^0 concentrations in air at ground level. The time modeled was a daytime period in August when the pond surface temperature was taken to be 32 to 34°C from 0900 to 1900 hours; the mean wind speed was measured as 0.27 m/sec ($\sim 0.6 \text{ mph}$); the prevailing wind was from the SW (see windrose given in Fig. 6); and only the largest of the two waste ponds (pond 1) was assumed to be a source of Hg^0 .

Fig. 6. Isopleths of Hg^0 concentrations expected at ground level given one waste pond as the primary Hg source; other assumptions as described in the text.

Since the model is not generally applied to short-distance dispersion of an air pollutant from an area source, the contributing pond must be approximated as a point source (at the pond center). Though the model does consider terrain effects, the influence of the ridge-valley province on the wind direction and hence atmospheric dispersion is obvious. Of most interest is the small areal extent of the $> 1000\text{-ng/m}^3$ concentration range and the rapid decrease in Hg concentration to estimated regional background levels within ~ 1.5 km of the center of the waste ponds even under meteorological conditions chosen to maximize the effect of the source area on local air quality.

The local deposition of Hg was also measured during the period of air monitoring. Duplicate bulk deposition collectors were situated 3 m from the pond edge to sample wet plus dryfall for a 68-h period (Fig. 5). During this time, one rain event was sampled which lasted approximately 1 h and resulted in 0.25 cm of precipitation. The concentrations of Hg in the bulk samples were 1.14 and 1.39 $\mu\text{g/liter}$. These levels are approximately an order of magnitude higher than concentrations of Hg in rain collected in Walker Branch Watershed, an area influenced by emissions from coal-fired power plants (Andren and Lindberg 1977), and ~ 25 times higher than Hg in bulk precipitation in a remote area of New Hampshire (Schlesinger et al. 1974). The deposition rates of Hg during this period, as calculated from the above two concentrations, were 7.4 and 9.0 $\mu\text{g/m}^2\text{d}$ which are from 15 to 20 times higher than the estimated deposition rate of Hg around five active Swedish chloralkali plants (Jernelov and Wallin 1973). However, in their calculations Jernelov and Wallin purposefully excluded the

area within a radius of 500 m of the plants because of the large variability associated with the data, stating only that the deposition rates in this area were "very high."

A useful relationship between deposition of an element and its air concentration has been introduced by Chamberlain (1953). Deposition velocity (V_g in cm/s) is an expression which relates depositional flux of an element (F_d in $\mu\text{g}/\text{cm}^2\text{s}$) to its mean air concentration ($\bar{X}$ in $\mu\text{g}/\text{m}^3$) as $V_g = F_d \cdot \bar{X}^{-1}$. Using the mean deposition of Hg during this period ($8.2 \pm 0.8 \mu\text{g}/\text{m}^2\text{d}$) and the mean air concentration ($991 \pm 2 \text{ ng}/\text{m}^3$; Table 5) yields a deposition velocity of $0.01 \text{ cm}/\text{sec}$. To our knowledge there are no other known estimates of V_g for Hg to any surface. Cawse and Peirson (1972), however, measured V_g for 29 other elements that averaged $\sim 0.38 \text{ cm}/\text{sec}$. Recently Cawse (1974) determined V_g 's for selected aerosol-associated elements over a two-year period at seven open-field locations using artificial surfaces and found a range of from 0.04 to 5.9 cm/s with spatial variations for any one element of $\pm 86\%$ of the mean.

Since Hg exists in the air near the ponds primarily as a gas, it is useful to compare the estimated V_g to those for other gaseous pollutants. Heinemann et al. (1976) report V_g for elemental iodine to grass as ranging from 0.3 to 2.7 cm/s while Unsworth and Fowler (1976) measured the deposition of SO_2 to a wheat canopy with V_g in the range 0.11 to 0.76 cm/s. Our limited data, combined with the fact that our measurements were taken very close to the source (above the ground) and with an artificial surface, demand that the calculated V_g be regarded cautiously. However, the possibility that Hg^0 may have a very

low deposition velocity near the source suggests its residence time in the atmosphere to be long. This emission of Hg^0 from this area may contribute more to a regional pollution problem than to a local one on a long time scale.

In addition to the data collected at the immediate pond site, we collected a limited number of soil and plant samples from an area generally downwind of the source. Samples of surface and subsurface (0.5-m depth) soils were collected from a housing development (Perrytown; see Fig. 6), approximately 0.8 km from the nearest waste pond. The sampled vegetation consisted of composites of fescue collected from small grassy knolls which were established on the edge of one waste pond. The soil concentrations of 0.085 $\mu\text{g/g}$ in surface soils and 0.072 $\mu\text{g/g}$ in subsurface soils do not appear to be elevated and compare favorably with the average earth crustal abundance of Hg of 0.07 $\mu\text{g/g}$ (Vinogradov 1959) and soils throughout the United States (Van Horn 1975). The levels of Hg in fescue growing directly in the waste pond (5.2 $\mu\text{g/g}$ in unwashed stems, 3.9 $\mu\text{g/g}$ in washed stems), however, are elevated by a factor of ~ 50 over world averages for other grasses (Van Horn 1975). The inability of thorough washing with distilled water and gentle rubbing to significantly decrease the Hg levels suggests the element to be largely internal to the plant tissue.

Aquatic Hg Emission

The location of the waste ponds behind steep dikes along a riverbank and perched on the flank of a ridge is highly conducive to the leaching of soluble components of the waste by direct rainfall and

by runoff from the adjacent ridge. Clay tiles installed beneath the dikes serve to drain the waste from the bottom, while overflow structures drain the ponds from the surface and intermediate depths. Discharge of waste leachate through these structures and many intermittent seepages flow directly into the adjacent river. Our own direct measurements of leaching rates of Hg from the waste area were limited to periods of very low river discharge, because most of the tile drains were submerged and thus inaccessible for sampling and flow measurement during higher flows. Consequently, we also applied a mass balance approach to estimate net fluxes of mercury to the adjacent river (Table 6). Using mercury concentrations and stream discharge measurements taken between 1974 and 1976 at points immediately upstream and downstream of the waste pond area (Turner and Lindberg 1978), we calculated net fluxes of mercury as the difference between upstream and downstream fluxes. To the extent that mercury accumulates in river bed sediments during periods of low river discharge, or is flushed from river bed sediments during periods of high river discharge, mass balance calculations based on instantaneous measurements will either underestimate (at low discharge) or overestimate (at high discharge) the actual leaching rate of mercury from the waste ponds. Direct measurements of all accessible flows of waste leachate into the adjacent river during a period of low river discharge offered some reasonable verification of a mass balance calculation made for the same period. On one day the total measured flux of mercury was ~ 14 g/d while the calculated net flux was ~ 28 g/d. The lower measured flux likely results from our inability to locate and measure all leachate

Table 6. Summary of calculated mercury fluxes from the chloralkali plant waste disposal pond area using the mass balance approach

Upstream ^a			Downstream ^b			
Total Hg ($\mu\text{g/liter}$)	Stream discharge ^c (m^3/s)	Total Hg flux ($\mu\text{g/day}$)	Total Hg ($\mu\text{g/liter}$)	Stream discharge ^c (m^3/s)	Total Hg flux (g/day)	Net Hg flux (g/d)
0.077	1.76	11.7	0.220	1.98	37.6	25.9
0.006	1.70	0.88	0.167	1.92	27.7	26.8
0.007	1.61	0.98	0.185	1.81	28.9	27.9
0.008	1.57	1.09	0.193	1.77	29.5	28.4 ^d
0.052	2.83	12.7	0.186	3.19	51.3	38.6
0.012	6.00	6.22	0.116	6.80	68.2	62.0
0.003	11.5	2.98	0.094	12.2	99.1	96.1
0.002	13.1	2.26	0.087	14.8	111	109
0.005	14.5	6.26	0.084	16.3	118	112
0.008	19.1	13.2	0.123	21.5	228	215
0.015	26.7	34.6	0.103	30.1	268	233
0.012	18.2	18.9	0.209	20.6	372	353

^a2.7 km above chloralkali plant.

^b6.4 km below chloralkali plant.

^cBased on discharge recorded at nearby U.S. Geological Survey stream gaging station and adjusted for difference in drainage area.

^dMeasured flux was 14.3 g/d.

seepages rather than from an overestimate by the mass balance calculation. By comparison, on one day while the plant was still in operation, the flux of mercury into the river was reported (USD0I 1970) to be 260 g/d.

Our estimates of aquatic fluxes of mercury from the waste pond area indicate considerable variability in the apparent leaching rate of mercury from the waste (Table 6), but demonstrate a good correlation (Fig. 7) with discharge of the adjacent river which responds primarily to local rainfall in the upstream 575-km² catchment. However, it is not clear whether this correlation results from a direct dependence of mercury-leaching rate on rainfall input or from the dependence of bed sediment erosion on river discharge. Both processes likely contribute to higher net mercury fluxes from the waste pond area during periods of higher river discharge.

Significance of Waste Pond Source

Using simple regression analyses (Figs. 4 and 7), we obtained functions which estimate both surface emission of Hg⁰, based on air temperature, and aquatic leaching of Hg, based on streamflow. Applying these relationships to daily streamflow and hourly temperature data for the waste pond area for a typical year (October 1973 - September 1974), we calculated the estimated annual loss of mercury from the waste disposal site by both the atmospheric and aquatic pathways. Losses were nearly equally divided between these sources, 36 ± 4 kg/year atmospheric emission of Hg and 39 ± 2 kg/year aquatic emission of Hg (predicted sum ± the estimated standard error of the sum of predicted

Fig. 7. Relationship between the net flux of mercury from the Saltville, Virginia, area (RK 2.7 to 6.4) to the discharge rate of the North Fork of the Holston River at RK 6.4. The simple correlation coefficient between these variables is $r = 0.86$ at $P < 0.01$. Deleting the one apparent outlier improves the correlation to $r = 0.98$ and permits prediction of net flux (ΔL) from the downstream NFHR discharge (Q_d) using the equation $\Delta L = a + b Q_d$, where $a = 10.8$ (standard error = 7.6) and $b = 7.6$ (standard error = 0.6).

fluxes). The occurrence of high streamflow during cold winter months resulted in predominance of the aquatic pathways during winter, while the atmospheric route predominated during summer months. This results in a relatively uniform emission of Hg from the waste deposits over an annual cycle.

The estimated total annual flux of Hg from this inactive plant, 75 ± 5 kg/year, is approximately 6% of the U.S. Environmental Protection Agency standard for emission of Hg to air and water from active chlorine plants situated in riverine environments. However, recently reported losses of Hg to the atmosphere and surface waters by Swedish chloralkali plants were in the range of 150 to 1400 kg/year (Wallace et al. 1971, Wallin 1976). Values for U.S. plants are estimated to range up to 1100 kg/year. Estimates of the capability of improved technology to reduce Hg emissions from active plants indicate a possible 40% decrease in emissions via these routes (Van Horn 1975), resulting in Hg losses from the more efficient operations (i.e., those reported above for Swedish plants) which would be similar to those reported here for waste disposal ponds alone.

The minimum quantity of mercury still remaining in the waste pond system can be estimated based on core data from one of the ponds. Mercury concentrations in the 2-m core from Pond 1 were found to be highest in the upper portion but a reasonable overall average would be ~ 100 $\mu\text{g/g}$ (Table 2). An estimate of the total mercury burden of the upper 2 m of this pond can thus be calculated. Using the surface area of this pond, 2.9×10^5 m^2 (66% of total surface area of both ponds), and assuming a waste density of 2.0 g/cm^3 leads to a waste

mass of $\sim 10^9$ kg. If the mercury content of this mass indeed averages 100 $\mu\text{g/g}$, then the total mercury burden of the upper 2 m of this pond is $\sim 9 \times 10^4$ kg. Assuming that less than 1% of this mercury is dissolved or readily soluble and using the above estimated annual leaching rate of mercury, we conclude that it would take on the order of 25 years to totally deplete the top 2-m of the pond of its soluble mercury burden. A similar calculation can be done for the volatilization of Hg^0 from the pond surface. Assuming the dispersed elemental Hg in the solid waste to be the primary source of volatile Hg, assuming only 10% of the Hg in the upper layer to exist as dispersed Hg^0 , and applying the above estimated annual emission rate of 36 kg/year of Hg^0 yields an approximate depletion time for Hg^0 in the upper pond layers of ~ 250 years. Since only the upper 2 m of this pond were considered as a source for mercury in this calculation, the estimate of flushing time is obviously minimal, with the upper 2 m comprising $< 10\%$ of the total waste volume.

Currently, the most significant mercury waste problem in this industry involves disposal of large quantities of solid waste materials. Thus, given the common disposal practice of ponding solid wastes, along with the residence time of Hg in existing large-scale waste deposits, it becomes apparent that the role of such residual waste deposits in local Hg cycles will continue to grow in relative importance.

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3. DISTRIBUTION AND BEHAVIOR OF MERCURY IN THE WATER AND SEDIMENTS OF THE HOLSTON RIVER-CHEROKEE RESERVOIR SYSTEM

Ralph R. Turner, Steven E. Lindberg, and John R. Lund

The work described in this chapter concerns investigation of the distribution of mercury within the river-reservoir system which for 20 years received effluents from the chloralkali plant located in Saltville, Virginia. The objectives of this part of the study were to determine (1) the levels of mercury which presently occur downstream of the plant site and the extent to which these levels are "relict" and thus not the result of current losses and (2) the natural processes which are regulating the downstream distribution of mercury and hence its ultimate fate in the receiving waters.

Study Area

The North Fork of the Holston River (NFHR) is a fast-flowing mountain stream which originates ~ 77 km above the former plant site (Fig. 8). The stream flows 133 km southwest through a rural landscape to the confluence with the South Fork of the Holston River at Kingsport, Tennessee. Thence, the main Holston River flows 50 km into Cherokee Lake, a 12,260-ha reservoir created in 1942 by the Tennessee Valley Authority.

The NFHR is characterized over most of its length by a coarse rocky substrate with extensive riffle and pool areas. Mean daily discharge (1920-1975) at a point 2.7 km upstream from the former plant site is $8.5 \text{ m}^3/\text{s}$ (drainage area = 575 km^2). Downstream near the

Fig. 8. Map of Holston River drainage system showing site of the inactive chloralkali plant and location of the major downstream impoundment, Cherokee Lake.

confluence with the South Fork the mean daily discharge (1931-1975) is $25.4 \text{ m}^3/\text{s}$ (drainage area = 1740 km^2) (U.S. Geological Survey 1975). The drainage area of the South Fork Holston River (SFHR) (5304 km^2) is three times that of the NFHR, and thus discharge below the confluence is dominated by the South Fork, the flow of which is regulated by several impoundments.

Throughout this chapter sampling locations within the river-reservoir system are referred to by river kilometer (RK) above or below the former chloralkali plant site. Table 7 gives the locations of all sampling sites using both the RK notation and the more familiar river mile notation.

The mercury cell chloralkali plant at Saltville, Virginia, was the only such plant ever located within the Holston River watershed. Additional anthropogenic sources of mercury, such as a hosiery mill, a pulp mill, and municipal sewage treatment plants, exist within this large watershed (below the confluence of the NFHR with the SFHR), but these were considered minor compared with the chloralkali plant. However, these sources likely contributed to the total mercury burden of water, suspended matter, and bottom sediments during this study and earlier.

Methods and Materials

Water Sampling Methods

Water samples were collected and filtered as described in Chapter 2. The concentration of mercury on the suspended matter ($\mu\text{g/g dry wt}$) was calculated as the ratio of the particulate mercury

Table 7. Location of sampling stations on the Holston-Cherokee System

Station description	Actual river mile	Miles above (-) and below plant	Kilometers above (-) and below plant	Drainage area (km ²)
North Fork Holston River				
Chatham Hill	109	-26.5	-43.0	
Old Broadford	91	-8.5	-14.0	
Neil Bridge	88.1	-5.6	-9.0	
McCrary	85.9	-3.4	-5.5	
USGS Station (0.4 mile below)	84.2	-1.7	-2.7	572
Chloralkali plant	82.5	0	0	
Transect	80.6	1.9	3.1	650
Below disposal ponds	81.0	2.3	3.7	
Railroad trestle	78.5	4.0	6.4	669
McKenna Island	76.5	6.0	9.7	750
Swinging bridge	72.0	10.5	17.0	769
Hayters Gap	69.5	13.0	21.0	875
Holston	59.5	23.0	37.0	1041
Hines Island	56.0	26.5	43.0	
Mendota	39	43.5	70.0	1276
Hilton	22	60.5	97.0	1432
Gate City USGS Station	8.8	73.7	119.0	1740
Kingsport	0	82.5	133.0	1886
Amis	110	114.5	184.0	
John Sevier Detention Lake	107	116.0	188.0	
Cherokee Reservoir				
H. B. Day Bridge	103	121.5	196.0	
Malinda	98	126.5	204.0	
Fall Creek dock	81	143.0	230.0	
Rt. 25E bridge	76	148.0	238.0	
Cherokee Dam	52	172.0	277.0	

concentration in water ($\mu\text{g/liter}$) to the total suspended matter concentration (mg/liter). Particulate mercury was generally determined as the difference between total and filtered mercury concentrations.

Table 8 summarizes results of analyses on replicate water samples collected upstream and downstream of the chloralkali plant site. Preliminary evaluation of the instantaneous variability in total and dissolved mercury across the direction of flow at RK 3.1 (5/21/75) indicated the need to composite several subsamples collected across the direction of flow at this site or to collect water samples further downstream where horizontal mixing of the river water and the settling pond effluent was more complete. Consequently, the first downstream sampling site was relocated to RK 6.4 and, as a further precaution, all water sampling at all sites consisted of compositing water collected across the direction of flow. Replicate samples collected in this manner at RK 2.7 and 6.4 and analyzed for total mercury (Table 8) indicated reasonable precision (average standard error upstream = 27%; averaged standard error downstream = 6%) in obtaining representative samples for total mercury. The poorer precision at the upstream location was obviously related to analytical imprecision because mercury concentrations at this site were close to our limit of detection.

An aliquot of only one of the two replicate samples collected at RK 2.7 and 6.4 (Table 8) was filtered and analyzed for dissolved mercury and total suspended matter. Thus we could not evaluate the precision in obtaining representative samples for dissolved mercury or mercury on the suspended matter (which is calculated from the

Table 8. Results of analyses on replicate water samples collected upstream and downstream of the chloralkali plant site

Location (RK)	Date	Time	Flow (m ³ /s)	Total Hg (μg/liter)	Dissolved Hg (μg/liter)	Total suspended matter (mg/liter)	Total suspended mercury ^a (μg/g)
-2.7	1/13/76	1500	6.00	0.012			8.0
-2.7	1/13/76	1500	6.00	0.011	<0.001	1.5	7.3
-2.7	1/14/76	1115	18.2	0.009	<0.001	62	0.14
-2.7	1/14/76	1115	18.2	0.014			0.22
-2.7	1/14/76	1815	19.1	0.008	<0.001	37	0.21
-2.7	1/14/76	1815	19.1	0.009			0.24
-2.7	1/15/76	1800	14.5	0.004			0.36
-2.7	1/15/76	1800	14.5	0.006	<0.001	11	0.54
-2.7	1/15/76	1800	14.5	0.004			0.36
-2.7	1/15/76	1800	14.5	0.005			0.45
-2.7	1/16/76	945	13.1	0.003	<0.001	4.8	0.63
-2.7	1/16/76	945	13.1	0.001			0.21
3.1S ^b	5/21/75	1720	13.0	0.064	0.034	4.1	7.3
3.1M ^b	5/21/75	1720	13.0	0.082	0.043	9.5	4.1
3.1N ^b	5/21/75	1720	13.0	0.240	0.104	8.7	16
3.1S ^b	7/30/75	900	1.71	0.193	0.064	6.6	20
3.1M ^b	7/30/75	900	1.71	0.199	0.072	10.0	13
3.1N ^b	7/30/75	900	1.71	0.155	0.064	9.2	10
6.4	1/13/76	1630	6.8	0.115	0.046	2.2	31
6.4	1/13/76	1630	6.8	0.117			32
6.4	1/14/76	1045	20.6	0.230	0.031	44	4.5
6.4	1/14/76	1045	20.6	0.186			3.5
6.4	1/14/76	1745	21.5	0.117	<0.001	53	2.2
6.4	1/14/76	1745	21.5	0.127			2.4
6.4	1/15/76	1700	16.3	-			-
6.4	1/15/76	1700	16.3	0.081	0.027	10	5.3
6.4	1/15/76	1700	16.3	0.087			5.9
6.4	1/15/76	1700	16.3	0.083			5.5
6.4	1/16/76	1015	14.8	0.085	0.052	5.8	5.7
6.4	1/16/76	1015	14.8	0.089			6.4

^aCalculated as ratio of particulate Hg (μg/liter) to total suspended matter (mg/liter).

^bS = south side of river, M = middle of river, N = north side of river; settling ponds are located on north bank of river.

difference between total and dissolved mercury and the total suspended matter concentration).

Sediment Sampling Methods

River sediment was collected at low water stages when most sampling sites could be waded. An effort was made to locate areas of recent local deposition of fine-grained sediment (clay and silt) within a few hundred meters of each site. These areas were usually behind anchored debris and in slow-moving pools. Collection was accomplished by gently scooping off the top centimeter or less of sediment and placing in a plastic "Whirl-Pac" bag or similar container. In deeper river water an Ekman dredge was used to collect sediment, with only the top centimeter being retained for analysis. At most sites, sediment was collected at two or more areas on opposite sides of the river, or in a transect across the direction of flow. Usually these replicate samples were composited before Hg analysis but on occasion were analyzed separately to determine lateral variability within a site. Table 9 gives the results of mercury analyses and loss on ignition on replicate samples from opposite sides of the river and indicates very good agreement in nearly every case. Sediment samples were stored on ice for return to the laboratory.

At the deeper sediment sampling sites within Cherokee Lake and John Sevier Detention Lake, a piston corer (5-cm ID) was used to obtain bottom sediment. Cores were extruded immediately and subsampled at 5-cm intervals from top to bottom.

Several published studies (cf. Thomas 1972, 1973; Cranston and Buckley 1972; Crecelius et al. 1975; Helmke et al. 1977) have shown

Table 9. Results of analyses of replicate river sediment samples collected on opposite sides of the NFHR in September 1975 (only fine-grained (silty-clayey) samples were collected)

Location (RK)	Total Hg ($\mu\text{g/g}$)	Loss on ignition (%)
-43	0.13	8.0
-43	0.14	9.9
-14	0.08	3.4 ^a
-14	0.16	10.2
-9	0.17	10.4
-9	0.16	10.6
3.7	20	5.5
3.7	18	6.3
3.7	18	7.8
9.7	9.4	10.9
9.7	9.1	9.6
17	6.0	6.6
17	6.3	8.3
21	4.2	7.3
21	4.4	6.8
43	3.0	6.5
43	1.9	6.2
133	0.96	5.6
133	0.88	6.9

^aSample contained some fine sand.

that mercury in river and lake sediments is associated predominantly with the smallest particle sizes which have the highest specific surface areas and also frequently the highest organic matter content. Coarser particles, especially sand-sized (0.062 - 2.0 mm) and larger, are usually composed of quartz and feldspar which are inert with respect to mercury and thus may be considered as diluents in sediment samples. In comparative studies of total mercury concentration in river and/or lake sediments, it is necessary to assure that samples submitted for Hg analyses are texturally similar. Judicious field collection biased towards collecting only fine-grained (clayey) sediments at all sites is one approach to achieving textural uniformity in a sample suite, but it is not always successful. We supplemented biased field collection with mechanical size separation of sediments in the laboratory. Such separation techniques should probably be used sparingly, and certainly very carefully, as they increase the chances of contamination by, or loss of, Hg in the laboratory.

Size-Fractionation Technique

Enough filtered NFHR water was added to each sediment sample to obtain a wet slurry which was stirred/agitated vigorously to disaggregate/homogenize the sediment. Sufficient homogenized sediment was poured off for Hg and other analyses on an unfractionated aliquot. More river water was added to the remaining sediment and the sample was stirred and partially poured into a clean, pre-wetted, sieve (stainless) of 44- μ m mesh size. By swirling the diluted slurry and allowing a few seconds for settling of coarser particles before pouring

into the sieve, much of the finer fraction was passed through the sieve prior to emptying the remaining sample into the sieve. Once all the sample was in the sieve, a wash bottle filled with river water was used to gently spray the finer particles through the sieve. This "wet sieving" was continued until the water passing through the sieve was clear. At this point the size separation was assumed to be complete, and the coarse fraction was washed out of the sieve with distilled water into a pre-weighed drying container which was placed in a drying oven. The fine fraction, greatly diluted by river water, was reconcentrated by centrifugation or simply by settling and decanting. When most of the excess water had been removed, the fine fraction was transferred quantitatively to a pre-weighed drying container and placed in a drying oven, set at $< 50^{\circ}\text{C}$. After drying of both fractions was complete (usually less than 24 h in a forced-air oven), the fine fraction was submitted for total Hg analyses. A few Hg analyses on unfractionated samples were run to check recoveries and the distribution of Hg between size fractions. Results of such checks are given in Tables 10 and 11.

Chemical Analyses

Laboratory analyses for mercury in both solid and aqueous samples were conducted as described in Chapter 2. Methylmercury (MeHg) was also determined in selected sediment samples using a combination of procedures given by Chau and Saitah (1973), Longbottom et al. (1973), and Bisogni and Lawrence (1974). Recoveries of MeHg from sediment were checked using ^{203}Hg -labelled MeHg. The recoveries ranged from 50 to

Table 10. Results of particle size fractionation and mercury analyses on sediments from core collected at RK 188 on Holston River

Interval (cm)	Percent < 44 μ m	Unfractionated total Hg (μ g/g)	44 μ m total Hg (μ g/g)	Fraction total Hg in < 44 μ m (%)
0-5	23	0.84	2.4	65
5-10	24	1.9	5.8	73
10-15	85	8.1	10.6	110
15-20	93	8.7	8.8	94
20-25	93	4.3	6.2	130
25-30	93	3.5	3.8	100
30-35	93	4.1	4.1	93
35-40	67	4.6	5.9	86
40-45	66	4.6	5.9	85
45-50	80	4.0	4.4	88
50-55	91	3.2	2.8	80
55-60	84	4.3	4.7	92
60-65	72	5.2	5.7	79
65-70	69	3.8	4.7	85
70-75	67	4.2	5.1	81
75-80	30	1.4	3.1	66
80-85	88	4.7	4.8	90
85-90	83	4.9	5.2	88
90-95	78	3.6	3.8	82
95-100	41	2.7	5.0	76

^aIndicates probable contamination of < 44- μ m fraction during wet sieving.

Table 11. Mercury concentrations in size-fractionated sediments from three tributaries of the Cherokee Reservoir

Location	Depth (cm)	Percent < 44 μ m	Σ % loss on ignition	< 44 μ m % loss on ignition	Hg (μ g/g)		
					Σ	< 44 μ m	> 44 μ m
German Creek ^a	0-5	40	2.7	7.4	0.146	0.290	0.050
	5-10	60	2.8	3.8	0.132	0.171	0.067
Galbraith Springs ^b	0-5	12	1.6	7.3	0.177	1.090	0.043
	5-10	12	1.1	7.4	0.152	0.890	0.052
Poor Valley Creek ^a	0-5	11	2.5	3.1	0.077	0.220	0.053
	5-10	50	3.5	5.1	0.111	0.135	0.061

^aSee Figure 9.^bAt RK (river kilometer) 216.

Fig. 9. Location of surface sediment and bottom core sampling stations in Cherokee Reservoir.

75%. Cesium-137 activity in lake sediment samples was determined using nondestructive gamma ray spectrometry (Ritchie et al. 1973, Pennington et al. 1973).

Results and Discussion

Distribution and Behavior of Mercury in River Water, Suspended Matter, and Bottom Sediment

Total mercury concentrations measured in river water immediately downstream of the plant site averaged 0.15 µg/liter during the study period (Table 12), while total mercury concentrations immediately upstream averaged 0.008 µg/liter. Essentially all the mercury in river water upstream of the plant site was in particulate form, but about one-third ($33 \pm 16\%$) of the mercury immediately downstream occurred in dissolved form. This is consistent with the solubility of the mercury in natural seepage from the waste disposal ponds. The concentration of mercury in suspended matter and bottom sediment likewise revealed sharp increases downstream ($\bar{x} = 7.6$ and 19.3 µg/g respectively) relative to upstream ($x = 0.41$ and 0.13 µg/g respectively). Thus, although there has been no production of chlorine since 1972, the observed 20-fold increase in total mercury concentrations in water and in suspended sediment demonstrates the presence of a large continuing loss of mercury from the vicinity of the plant. The 200-fold increase in bottom sediment concentrations probably reflects local deposition (by sedimentation) of mercury-contaminated solids immediately downstream of the waste disposal site, but it could also result from an inadvertent sampling of "relict" sediment deposited while the plant was still in

Table 12. Mean ($\bar{X}$) and standard deviation (S.D.) of mercury concentration in water, suspended matter, and bed sediment immediately upstream and downstream from the former chloralkali plant site during the study period.

		Water		Suspended matter Hg ($\mu\text{g/g}$)	Bottom sediment ^a Hg ($\mu\text{g/g}$)
		Total Hg ($\mu\text{g/liter}$)	Dissolved Hg ($\mu\text{g/liter}$)		
Upstream -2.7 km	$\bar{X}$	0.008	<0.001	0.41	0.13
	S.D.	0.004	—	0.17	0.03
	N	10	9	7	7
Downstream 3.1 to 6.4 km	$\bar{X}$	0.15	0.05	7.6	19.3
	S.D.	0.05	0.02	3.8	1.2
	N	11	11	10	3

^aSilt-clay fraction only.

operation and presumably discharging higher levels of mercury-containing wastes. Although our sampling strategy of collecting only fine-grained, easily resuspended, unconsolidated surface sediment was specifically directed at avoiding such "relict" sediment, we cannot dismiss this possibility.

Since the natural seepage from the waste disposal area did not contain mercury in particulate form, the occurrence of mercury (largely in the particulate form) immediately downstream of the waste disposal area indicated rapid adsorption of dissolved mercury onto the suspended matter during mixing of seepage and river water. In view of the strong affinity of mercury species in solution for natural suspended matter, as reported by many others (Cranston and Buckley 1972, Lindberg et al. 1975, de Groot et al. 1971, Jenne 1970), this rapid shift in phase was not unexpected.

It is instructive to calculate the resulting mercury concentration on suspended matter in the river immediately downstream of the source area if all of the mercury estimated to be lost annually from the waste disposal area (39 kg/year, see Chapter 2) is assumed to be rapidly adsorbed onto the suspended matter. We estimate that the annual total suspended matter flux past the former plant site is $\sim 1.3 \times 10^7$ kg (given a weighted average suspended matter concentration of 50 mg/liter and the average annual stream discharge of $2.6 \times 10^8 \text{ m}^3$). If the annual loss of mercury were divided homogeneously over this annual suspended matter load, then the resulting mercury concentration on the suspended matter would be about 2.8 $\mu\text{g/g}$. This value is less than one-half the average value observed here (Table 12). Thus, we

hypothesize that the suspended matter load of the NFHR is contaminated to higher levels during low river flow when the relative impact of the waste leachate on downstream mercury levels in water and suspended matter is greatest. In support of this hypothesis, stream discharge and mercury concentration on suspended matter immediately downstream of the plant site are inversely correlated ($r = -0.86$, $P > 0.05$). The plot of these variables (Fig. 10) suggests that the relationship is not linear. Significantly, mercury concentrations on suspended matter increased systematically with decreasing stream discharge during the falling stage of a storm hydrograph (Fig. 10), i.e., with increasing relative influence of the point source of mercury.

Variations in mercury concentrations in water, suspended matter, and bottom sediment over more than 250 km of the NFHR and its receiving waters are illustrated in Fig. 11. The location of the former chloralkali plant at river kilometer 0 (RK 0) is conspicuous by the peak concentrations of mercury observed immediately downstream from the plant site. Concentrations decrease rapidly downstream from the plant site to about RK 40, below which concentrations vary only slightly. Except as indicated, the data plotted in Fig. 11 represent results of a single synoptic (August 1975) survey of mercury in the river-reservoir system. As noted in Table 12, for most of the river stations immediately upstream and immediately downstream from the plant site, there was considerable temporal variability in concentrations.

Bottom sediments should, to some extent, integrate the variability in water and suspended matter concentrations and thus may reflect average conditions at a given station. Thus, by way of comparison, we

Fig. 10. Relationship between NFHR discharge and Hg concentrations on river suspended matter at 3.1 to 6.4 km below the plant site. Solid circles represent samples collected in May-August, 1975.

Fig. 11. Downstream distribution of dissolved and total Hg in the river water column (11a), total Hg on the river suspended matter (11b), and total Hg in the silt-clay fraction of river bottom sediments (11c). Reference 5 is Hildebrand et al. (1975).

have included bottom sediment data in Fig. 11c from an earlier (1973-74) study of mercury in this river system (Hildebrand et al. 1975). In general these distributions of mercury in bottom sediment as a function of river kilometer are similar. However, our observations of bottom sediment mercury concentrations below the chloralkali plant are somewhat lower than those reported earlier, perhaps reflecting a gradual flushing of the more highly Hg-contaminated sediments from the river system since plant closing.

The downstream distribution of total mercury in water lends itself to the application of a simple dilution model because the plant site represents a strong point source of mercury to the NFHR. The concentration of total mercury in water downstream from the source should be regulated primarily by dilution, sedimentation, vapor exchange with the atmosphere, and exchange with bottom sediments. All of these processes are normally expected to result in a downstream reduction in mercury concentration. We have sufficient information to quantitatively evaluate only the effect of dilution. Given an assumption of steady-state conditions (including constant source strength, a time-invariant downstream transport rate and complete physical mixing), the effect of dilution on mercury concentration at any downstream point can be estimated using the relative increase in drainage area as an index of the increase in streamflow rate at that point. Assumption of steady-state conditions is probably justified only over short periods of time (on the order of days) when streamflow and source strength are not rapidly changing. If only dilution is

operative, the concentration of mercury, C , expected at a downstream point with drainage area, A , will be

$$C = \frac{A_s C_s + (A - A_s) C_b}{A} ,$$

where A_s is the drainage area at any well mixed point below the source, C_s is the mercury concentration at A_s , and C_b is the regional background concentration of mercury (taken here to be 0.008 $\mu\text{g/liter}$, see Table 12). Basically the equation states that mercury concentration decreases downstream from a strong point source as a function of the increase in drainage area and in the concentration of mercury in the uncontaminated tributaries which account for the drainage area increase. The equation is derived from simple mass balance considerations such as outlined by O'Connor (1976) and should only be applied if streamflow rate is directly proportional to drainage area. Hydrologic data (U.S. Geological Survey 1975) for the two stream gauging stations (at RK -2.7 and 119) on the NFHR where the drainage areas are 575 km^2 and 1740 km^2 indicate that average daily stream discharge at these stations differs by a factor (2.99) which is nearly equal to the ratio of the drainage areas (3.03). Thus between RK -2.7 and 119 the above equation appears to be applicable.

Using the relationship between drainage area and river kilometer (Fig. 12) along with mercury concentrations observed immediately below the plant site on five different occasions (three from earlier studies, Hildebrand et al. 1975), we calculated mercury concentrations expected downstream from the plant due to dilution. Comparison of the observed

and expected concentrations of total Hg as a function of distance from the point source (illustrated in Fig. 13 for the five different occasions) reveals that a significant fraction of the initial rapid downstream decrease in mercury concentrations can be accounted for by dilution alone. Observed concentrations were generally greater than 40% of the predicted concentrations. The equation does not predict a uniform change in concentration as a function of distance downstream due to the stepwise increase in drainage area (Fig. 12) at each confluence of a tributary with the NFHR. Contributions from three tributaries within 20 km of the plant site increase the drainage area, and thus streamflow, by a factor of 1.3. Mercury concentrations observed 20 km below the plant site decreased by a factor which varied between 1.3 (Fig. 13a) and 3.1 (Fig. 13b). Significantly, the best agreement between the dilution model and field observations occurred for the observation period with the highest stream discharge (Fig. 13a), suggesting that the relative importance of processes other than dilution is dependent on flow conditions. To the extent that the travel time of a discrete water parcel between two points along the river is reduced at higher stream discharge (when dispersion is greatest), opportunities for losses of mercury from a water parcel by sedimentation and other time-dependent processes will also be reduced. Thus, the dilution model should indeed predict downstream mercury concentrations more successfully at higher stream discharge than at low discharge. The data for May 1974 (Fig. 13a) indicate that the dilution model can also underestimate downstream total mercury concentrations. Obviously, at higher and, hence, at more variable

Fig. 12. Change in drainage area as a function of river kilometer for the NFHR below the chloralkali plant site.

Fig. 13. Comparison of expected concentrations of total Hg in downstream water calculated from application of the dilution model with field observations under various stream discharge conditions.

stream discharges, the assumption of a time-invariant transport rate is less appropriate. Under such conditions higher total mercury concentrations in the water column arising from resuspended contaminated bottom sediment downstream from the point source could conceivably exceed concentrations predicted by the simple dilution model because an additional "internal" source is involved.

Application of this dilution model indicates that substantial quantities of mercury (up to 80% of that released at the plant site) may accumulate in downstream river sediment or are lost to the atmosphere via volatilization during low river discharge. This latter pathway would involve reduction of Hg^{++} to Hg^0 , a reaction shown to be bacterially mediated (Summers and Silver 1972). This should be relatively unimportant, however, in this well-oxygenated, cation-rich environment and over the short time scale involved here. Sediment deposited downstream under low discharge conditions can be resuspended during storm flows to produce higher concentrations downstream than would be expected from simple dilution of the point source, such as the case in Fig. 13a.

Alternatively, Hg incorporated into bottom sediments may be chemically or biologically methylated followed by volatile loss or biotic uptake. These pathways would result in a net loss of Hg from the downstream water/sediment system. Analysis of four surface sediments collected downstream from the source area revealed very low levels of methylmercury. The concentrations ranged from 2100 to 3300 ng/g total Hg and 0.2 to 1.3 ng/g methylmercury. The indication that 0.02 to 0.05% of the total Hg present is in the methyl form agrees

with previously reported results for both freshwater and marine sediments (Andren and Harriss 1973, Batti et al. 1975). Hildebrand et al. (1975) (see also Chapter 4) have shown that, in the NFHR, a significant fraction of both the fish burden of Hg (75 - 90% of total) and the benthic invertebrate burden of Hg (29 - 83%) is in the methyl form. Thus, the possibility exists that Hg methylated in the sedimentary environment is rapidly assimilated by benthic invertebrates and, ultimately, by fish and thus temporarily removed from the abiotic system.

Distribution of Mercury in Reservoir Sediments

Mercury in the silt-clay fraction of the surficial sediments of the receiving water body, Cherokee Lake, ranged from 0.47 to 2.40 $\mu\text{g/g}$. Although no detailed areal mapping of sediment mercury concentrations was conducted on the lake, the silt-clay fractions of surface samples collected at four points along the central axis of the lake (Fig. 9) revealed the highest concentration (2.70 $\mu\text{g/g}$ at RK 188) near the input of the Holston River and the lowest concentration (0.51 ± 0.05 $\mu\text{g/g}$ at RK 277) near the lake outflow. Thus, the decreasing gradient in mercury concentration in river surface sediments which characterized the environment below the former plant site was continued through the lake.

To establish the background mercury concentration expected in the lake sediment, additional sediment samples were collected in two semi-isolated coves receiving sediment from adjacent, unimpacted, tributary areas (German Creek and Poor Valley Creek, see Fig. 9).

Samples from shallow deltaic areas in these coves contained considerable material coarser than 44 μm in size (60 - 90% of total) and thus were size-fractionated prior to analysis for mercury. The silt-clay fraction of these cove sediments contained 0.14 to 0.29 $\mu\text{g Hg/g}$, or about one-fifth the levels observed in midlake sediments of comparable particle size and about the same levels as those found in the silt-clay fraction of river sediments upstream from the former plant site on the NFHR. Interestingly, the silt-clay fractions of sediment collected in the Poor Valley Creek and German Creek tributaries above any influence by Cherokee Lake contained 0.05 to 0.11 $\mu\text{g Hg/g}$, suggesting some contribution of NFHR-derived sediment to the cove sediments. Shacklette et al. (1971) give 0.147 $\mu\text{g Hg/g}$ as the average mercury concentration in uncontaminated soils and other surficial materials from the eastern conterminous United States. This value provides an appropriate comparison here only to the extent that mercury concentrations in river sediments reflect mercury concentrations in sediment source materials (soils and rocks). Thus in the absence of anthropogenic inputs of mercury to Cherokee Lake, we might expect that sediment levels should not exceed about 0.15 $\mu\text{g Hg/g}$, other factors such as the content of organic matter (Lindberg and Harriss 1974, Andersson 1967, Kennedy et al. 1971) also being considered.

Mercury distributions in clayey sediment cores from each end (RK 188 and 277) and the middle (RK 230) of Cherokee Lake suggested that the anthropogenic mercury loading of the lake has varied considerably with time (Fig. 14). Although the cores came from areas of the lake

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Fig. 14. Distribution of total Hg and ^{137}Cs in sediment cores from Cherokee Reservoir.

contrasting considerably in water depth and proximity to the major river input, the mercury concentration profiles have some similar features, i.e., the occurrence of highest concentrations at from 10 to 35 cm below the present sediment surface and a decreasing trend in mercury concentration in the most recently deposited sediments.

Cesium-137 activity profiles were used as approximate guides to the chronology of sediment deposition (Ritchie et al. 1973, Pennington et al. 1973), the peak atmospheric fallout and the hence the lake deposition of ^{137}Cs having occurred in 1963 (Robbins and Edgington 1975). A well-defined peak in ^{137}Cs activity occurred in only one core, that from RK 277, and thus provides the only reliable chronology.

Significantly, four attempts to obtain a core longer than 50 cm at RK 277 failed. All four cores, ranging in length from 43 to 50 cm, bottomed out in material more similar in appearance and texture to natural soil than to lacustrine sediment. We interpret this observation to mean that the 47-cm core represents the entire period (~ 34 years) of sediment deposition at RK 277. The fact that mercury concentrations decrease rapidly to background levels ($0.1 \mu\text{g/g}$) towards the bottom of this core is consistent with our interpretation, since the dam which impounded the Holston River was completed 10 years prior to the existence of the mercury-cell chloralkali plant within the drainage basin of the lake. Occurrence of the peak ^{137}Cs activity at 20 to 25 cm indicates that deposition of this layer occurred ca. 1963 and yields an average rate of sedimentation of $\sim 1.9 \text{ cm/year}$ for this site (RK 277). Extrapolating this rate over the total thickness ($\sim 50 \text{ cm}$) of sediment at this site yields a date for the deposition of

the oldest sediments of ca. 1949 which is reasonably consistent with the 1942 impoundment of the river.

Applying these estimated rates of sedimentation to the mercury profile in the core from RK 277 and assuming (1) no significant post-depositional redistribution of sediment-bound mercury and (2) a constant yearly rate of sedimentation indicates that the peak concentration of mercury occurred ca. 1958. However, the 5-cm subsampling interval may superimpose a 2- to 6-year uncertainty in this chronology. We have no explanation as to why mercury deposition at RK 277 should have been higher in the late 1950's than during other times, unless early operations of the chloralkali plant were especially inefficient with respect to aquatic losses of mercury.

The mercury and ^{137}Cs activity profiles in the cores from RK 230 and 188 are difficult to interpret. The slightly increasing trend in ^{137}Cs activity towards the bottom of these cores suggests that they were of insufficient length to penetrate into sediments deposited prior to 1963. This suggestion is consistent with an expected higher rate of sedimentation nearer to the major hydrologic input to the lake. Rates of sedimentation at RK 188 and 230 would thus have to be equal to, or greater than, 6.7 and 4 cm/year, respectively, in order to account for the burial of the 1963 layer by at least 100 and 60 cm of sediment, respectively. These are rather high rates of sedimentation compared to those for natural lakes but are well within the range reported for several impoundments in the United States (Ritchie et al. 1973). Average hydraulic flushing time (inflow/volume) for Cherokee Lake is about six months, and thus the annual sediment loading is potentially large.

The mercury peaks in the cores from RK 188 and 230, at 10 to 15 cm and 20 to 25 cm respectively, appear to be interrelated to each other but unrelated to the peak in the core from RK 277, since they probably represent sedimentary layers deposited since 1970. Regardless of the precise date of deposition of the highest mercury layers in these cores, these peaks must have occurred near the end of operations at the chloralkali plant since (1) there has been a steady rapid post-peak decrease (8.1 to 2.4 $\mu\text{g Hg/g}$ and 10.6 to 2.4 $\mu\text{g Hg/g}$) in concentration and (2) concentrations (2.8 to 5.7 $\mu\text{g Hg/g}$) below the peaks are consistently above background levels for this area. Any bottom scouring at these shallower coring sites would, of course, invalidate our interpretation of the ^{137}Cs activity profiles. Such scouring seems unlikely at RK 230 considering the large cross-sectional area of the lake at this site. The occurrence of several layers in the core from RK 188 containing considerable material $> 44 \mu\text{m}$ in size (see Table 10) strongly suggests that sedimentation rate at this site has been highly variable.

Post-depositional redistribution of mercury and ^{137}Cs and highly variable annual rates of sedimentation would complicate our interpretation of the vertical distribution of these constituents in the Cherokee Lake sediment cores. However, significant post-depositional redistribution of mercury in profundal lake sediments by processes other than physical mixing and bioturbation have not been reported (Lindberg et al. 1975, Bothner and Carpenter 1973). In reference to ^{137}Cs distributions in sediment cores, Krishnaswami et al. (1971) have pointed out that diffusion cannot shift or completely

obliterate subsurface peaks in concentration. Continuous physical mixing and bioturbation of surface sediments are likely to have been important at RK 230 and 188 due to the shallower water depth (< 10 m) and smaller lake cross-sectional area compared with RK 277 which is both deeper (35 m) and subjected to anaerobic conditions in bottom water. However, since these processes would be expected to affect the ^{137}Cs and mercury profiles equally, by downward mixing of the peaks (Robbins and Edgington 1975, Krishnaswami et al. 1971), they would not change the relative chronology of mercury deposition inferred from the ^{137}Cs profile.

Although the sediment data for Cherokee Lake are sketchy and somewhat ambiguous with respect to the chronology of earlier mercury loadings, it does appear that the lake has served as a major sink for mercury. Mercury levels in surface sediments are generally decreasing with time, possibly in response to a gradual decrease in both the source strength at the plant site on the NFHR and the continuing input of river sediment contaminated with mercury during plant operation.

4. DISTRIBUTION OF MERCURY IN FISH AND INVERTEBRATES OF THE NORTH FORK HOLSTON RIVER-CHEROKEE RESERVOIR SYSTEM

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INTRODUCTION

A considerable research effort over the last decade in many countries has been devoted to investigations on the distribution and bioaccumulation of mercury in aquatic systems. One of the major justifications for studying the cycling of mercury in aquatic systems is the potential threat to human health through consumption of methylmercury-contaminated aquatic resources (mainly fish and shellfish). A postulated sequence of events whereby aquatic biota can accumulate methylmercury in the absence of direct methylmercury discharge to waters includes the following (Jensen and Jernelov 1969, Wood et al. 1968, Jernelov 1968, Lander 1971):

(1) Mercury is discharged as elemental mercury or as mercury (II) chloride and hydroxide complexes.

(2) This mercury is incorporated onto suspended particulates and into the sediments within a short distance from the outfall.

(3) In the sediments, bacterial action converts the mercury to methylated forms. In basic waters dimethylmercury can be volatilized. In acidic waters any dimethylmercury is converted to the much less volatile monomethylmercury ion which is retained in solution.

(4) Monomethylmercury ion can be accumulated directly from the water by the biota.

(5) Predators can receive methylmercury both from water and from the food chain.

The Food and Drug Administration (FDA) "Action Level" for total mercury in fish and shellfish prior to 1978 was 0.5 $\mu\text{g/g}$. Since June 1978, the FDA "Action Level" for total mercury has been raised to 1.0 $\mu\text{g/g}$ (Federal Register, Vol. 44, No. 14). All references to the FDA "Action Level" in the following report address the 0.5- $\mu\text{g/g}$ level effective through June 1978.

Total mercury levels (above the 0.5- $\mu\text{g/g}$ FDA action level) in fish species from the North Fork Holston River were first observed in 1970, and subsequent investigations have verified the extent of total mercury contamination (Bailey 1974; Tennessee Valley Authority 1972, 1973, 1974; Milligan and Ruane 1978). Methylmercury levels in biota were not included in the studies prior to 1974. We initiated the biological phase of the Holston-Cherokee study: (1) to quantify the extent of both total mercury and methylmercury in fish and benthic invertebrates in the Holston-Cherokee system, and (2) to utilize the information collected in the physicochemical phase of this study (Chapter 3) to examine factors in the abiotic environment that affect mercury concentration in fish and invertebrates.

Study Area

The study area has been described in Chapter 3 of this report and the reader is referred to this section for a characterization of the Holston-Cherokee system. The biological phase of the work included eight sampling stations on the North Fork of the Holston River and

three stations on Cherokee Reservoir (Fig. 8). North Fork Holston River stations sampled were Chatham Hill, Neil Bridge, and McCrady (above the chloralkali plant), and below disposal ponds, McKenna Island, Hayters Gap, Hines Island, and Kingsport (below the chloralkali plant) (Table 7). Cherokee Reservoir stations sampled were immediately downstream from the John Sevier detention lake to the H. B. Day Bridge, the Rt. 25E bridge, and Cherokee Dam (Table 7).

Methods

North Fork Holston River

Fish and benthic invertebrates were collected at four stations in August 1974 (McCrady, below disposal ponds, Hayters Gap, and Kingsport), and at seven stations in August 1975 (Chatham Hill, Neil Bridge, below disposal ponds, McKenna Island, Hayters Gap, Hines Island, and Kingsport) (Table 7). We attempted to collect the following fish species at each station: rock bass (Ambloplites rupestris), hog sucker (Hypentelium nigricans), and shiners (Notropis spp.). Fish were collected by electrofishing in 1974 and by selective poison (rotenone) in 1975. Where possible, a wide size range of fish of each species was selected for mercury analysis. Fish were frozen immediately in the field on dry ice for transport to the laboratory. Total weight measurements were obtained for each fish analyzed.

Benthic invertebrates were collected at the same time and stations as fish. Invertebrates were collected qualitatively by physically disrupting the river substrate upstream from a collecting net. Specimens were frozen on dry ice immediately after collection.

Cherokee Reservoir

Moribund or dead striped bass (Morone saxatilis) were collected by personnel of the Tennessee Game and Fish Commission in June 1972 in the vicinity of the Center Hill boat dock. In January, April, and August 1973, healthy striped bass were collected by the same group near the same location by gill netting.

Other fish species were collected at three stations in August 1974 (immediately below John Sevier detention pond to the H. B. Day Bridge, Rt. 25E bridge, Cherokee Dam) and at two stations in August 1975 (immediately below the detention pond to the H. B. Day Bridge, Cherokee Dam). We attempted to collect the following fish species at each station: largemouth bass (Micropterus salmoides), bluegill (Lepomis macrochirus), gizzard shad (Dorosoma cepedianum), threadfin shad (Dorosoma petenense), and smallmouth bass (Micropterus dolmieu). In addition, striped bass (Morone saxatilis) were collected at Cherokee Dam (near Black Oak Dock) in 1975. All fish except striped bass were collected by electrofishing. Striped bass were collected in 1974 by hook and line. All fish were frozen immediately on dry ice for transport to the laboratory. Benthic invertebrates were not collected in Cherokee Reservoir. As in the North Fork, a wide size range of fish from Cherokee Reservoir were collected for mercury analysis. Total weight measurements were obtained for each fish analyzed.

Mercury Determinations in Fish and Invertebrates

Fish axial muscle and benthic invertebrate samples were analyzed for total mercury by the flameless atomic absorption method.

In the early stages of this work, a slurry of fish muscle was prepared by adding a measured amount of water to a weighed test portion of fish muscle. Generally two to three times as much water was added to 1 to 10 g of muscle and the mixture ground into a slurry in a blender of appropriate size.¹ After blending, test portions of the slurry were reserved for the determination of their total and methylmercury content. Since the slurry had a tendency to separate into a tissue and water phase on standing, test portions for the various analyses were removed immediately after preparation of the slurry.

In all subsequent work, the procedure was modified to eliminate the preparation of large amounts of slurry. Except in those cases where the amount of sample available for analysis was limited to approximately a gram or less, the sample was first divided into test portions for the various analyses, and slurries were prepared only of the portions used for the methylmercury analyses. The separation of methylmercury from the fish tissue or invertebrate tissue and subsequent extraction as methylmercury halide into benzene was accomplished more readily from a slurry than from chunks or small pieces of tissue.

The cold-vapor atomic absorption method for the determination of total mercury depends on the destruction of all organic material and

¹Blenders, working sizes; micro 3-20 ml, semi-micro 25-250 ml, macro 75-500 ml. Eferbach Corporation, P. O. Box 1024, Ann Arbor, Michigan 48106.

the oxidation of all mercury compounds to inorganic mercury (II). Smith's perchloric-nitric acid procedure (Smith 1957), as modified by Feldman (1974), was used to convert all samples (muscle, slurry, or invertebrates) to a solution of inorganic mercury (II). This solution was then diluted to a volume of 50 or 250 ml (depending on the size of sample taken for analysis) with demineralized water. The mercury content of this solution was then analyzed by the cold-vapor atomic absorption method.

Selected fish species and benthic invertebrates at selected stations were also analyzed for methylmercury in axial muscle (fish) or whole animal (invertebrates) utilizing a gas chromatograph equipped with a microwave emission spectrometric detector. The methylmercury in fish muscle or invertebrates was first converted to methylmercury bromide, and this compound was then extracted into benzene. A portion of the benzene extract was used for the methylmercury analysis.

A test portion of the fish slurry, containing 0.2 to 2.0 g of muscle or 0.2 to 2.0 g of ground invertebrate was weighed into a 20-ml screw cap vial. About 2 ml of demineralized water was then added to the invertebrates (the fish slurries contained enough water), and finally 1 ml of 1 M CuSO_4 solution (250 g $\text{CuSO}_4 \cdot 5 \text{H}_2\text{O}$ /liter of H_2O) and 1 ml of 3 M NaBr in 2 M H_2SO_4 (310 g NaBr/liter of 2 M H_2SO_4) were added. The vial was closed with a cap containing a polyvinyl liner and shaken for 15 min to mix the sample and reagents. Next, 2 ml of benzene was added and the mixture shaken for 15 min. The vials were centrifuged, 5 to 15 min, to aid in the separation of the phases, and some of the benzene was removed and stored in a vial

containing about 25 mg of anhydrous sodium sulfate. These vials were capped with teflon-lined diaphragms and stored in the dark until ready for methylmercury analysis by use of the G.C.-microwave emission system (Talmi 1975).

Benthic invertebrate samples were partially thawed in the laboratory, separated into taxonomic groups, then refrozen for latter analysis. When necessary, several taxa were combined to provide a sample large enough for chemical analysis. For the larger sized invertebrate taxa collected (Megaloptera, Decapoda), a replicate consisted of 1 to 10 individuals. For the smaller sized taxa (Hydropsychidae, Atherix, combined taxa), a replicate consisted of from 25 to 250 individuals. The use of blenders to homogenize samples of invertebrates was not successful, due to the limited amount of sample and/or the inability of the blender to sufficiently subdivide some of the insects external parts (thorax, legs, etc.).

Samples of invertebrates were ground to a very fine powder (50 to 100 mesh), at liquid nitrogen temperature, in a Spex No. 6700 Freezer Mill.² This grinding was accomplished by placing the sample in a polycarbonate tube (20-mm diameter x 100-mm length) containing a rod-pestle, and the ends of the tube then closed with magnetic end caps. The sample thus contained was placed in the freezer mill's liquid nitrogen bath. After closing the cover of the mill, the coils of the mill were alternately energized by solid state circuitry which drove the rod-pestle back and forth between the end caps, at up to 30

²Spex Industries, Inc., Box 798, Metuchen, N. J. 08840.

times per second. This action by the rod-pestle ground the frozen sample to a fine powder. Since no local heating occurred, decomposition of temperature sensitive compounds was averted. This finely ground powder yielded a moist paste on warming to room temperature. Portions of the ground samples were used for the total mercury and methylmercury analyses.

Analysis of Factors Affecting Concentration of Total Mercury in Fish and Invertebrates

The relationship of total mercury concentration to fish weight for selected species at all stations on the North Fork of the Holston River sampled was examined by linear regression analysis.

Concurrent sampling of fish, invertebrates, water, and sediments (Chapter 3, Fulkerson et al. 1974; Van Hook and Shults 1976) for total mercury concentration provided an opportunity to examine factors in the environment that can account for total mercury levels in fish and invertebrates. The North Fork of the Holston River from above the chloralkali plant down to Kingsport (Fig. 8) provided a range of mercury concentrations in water, sediment, and biota in similar habitats amenable to a multivariate analysis. By combining the 1974 and 1975 biological collections, 11 station-date combinations were available for analysis for which concurrent samples of total mercury concentration in water and sediment were available.

Factors affecting the concentration of total mercury in rock bass and hog sucker in the North Fork Holston River were examined using a backward elimination multiple regression procedure (Draper and Smith 1966, p. 167), as implemented in the Statistical Analysis System (Barr

et al. 1976). Independent variables included in this analysis were total mercury concentration in water (particulate and dissolved), dissolved mercury concentration in water, total mercury concentration in the < 44- μ m sediment size fraction, and mean total mercury concentration of representative benthic invertebrate taxa. The < 44- μ m size fraction of sediment was used in the analysis to achieve uniformity between stations and because the majority of total mercury in whole sediments in the North Fork is contained within this size fraction (Table 10). The dependent variable was total mercury concentration in axial muscle of rock bass or hog sucker, adjusted to a size of 100 g for hog sucker or 50 g for rock bass where a significant relationship of mercury concentration and fish weight was observed. If, for a particular station-date combination, the relationship of Hg concentration to weight was not significant ($P = 0.05$), the mean Hg concentration in fish muscle was used as the dependent variable. A similar multiple regression analysis utilizing the same 11 station-date combinations was completed with the dependent variable being total mercury concentration in benthic invertebrates and the independent variables being total mercury concentration in water (particulate and dissolved) and total mercury concentration in < 44- μ m sediment.

Results

The biological phase of the Holston-Cherokee study will be discussed separately for two regions: North Fork Holston River and Cherokee Reservoir.

North Fork Holston River

A total of 281 fish collected from the North Fork Holston River in 1974 and 1975 were analyzed for total mercury concentration in axial muscle. A total of 119 of these 281 fish were also analyzed for methylmercury. Rock bass, hog suckers (Fig. 15), and shiners all exhibited a similar general pattern in total mercury concentration, reflecting collection points above and below the chloralkali plant (Tables 13, 14). With the exception of rock bass collected in 1974 at McCrady (5.5 km above the plant), mean total mercury concentration in the three species examined in both years above the plant were below the FDA "action level" of 0.5 $\mu\text{g/g}$. Mean total mercury concentrations above the plant ranged from 0.121 $\mu\text{g/g}$ in shiners to 0.464 $\mu\text{g/g}$ in hog suckers. The mean total mercury concentration in rock bass collected in 1974 at McCrady 5.5 km above the plant was 0.704 $\mu\text{g/g}$ (Tables 13 and 14).

Fish species collected immediately below the chloralkali plant exhibited mean total mercury concentrations well above the FDA "Action Level." The mercury concentration then generally decreased downstream in all species. With the exception of hog sucker collected in 1974, all species still exhibited mean mercury concentrations above 0.5 $\mu\text{g/g}$ at Kingsport 133 km below the plant (Tables 13 and 14).

Methylmercury analysis of rock bass, hog sucker, and shiners collected both above and below the plant indicates that the major fraction of total mercury in fish muscle in the North Fork is methylmercury. Estimates of the mean percentage methylmercury of total mercury (Table 15) in the three fish species range from 78.2% (shiners

Fig. 15. Mean total mercury concentration in rock bass and hog sucker above and below the chloralkali plant. "Action Level" is FDA limit of 0.5 µg/g. Data from Table 13 and Table 14.

Table 13. Total mercury concentration in axial muscle (except where noted) of 3 fish species collected in the North Fork of the Holston River, August 1974. All total mercury values are $\mu\text{g/g}$ wet wt

Species ^a	McCrary 5.5 km above plant	Downstream disposal ponds 3.7 km below plant	Hayters Gap 21.0 km below plant	Kingsport 133 km below plant
RB				
n	13	11	16	10
$\bar{X}$	0.704	1.594	1.137	1.261
S.D.	0.408	0.546	0.389	0.806
Size range (g)	8-303	20-184	4-96	24-157
HS				
n	11	3	6	2
$\bar{X}$	0.464	1.420	1.235	0.427
S.D.	0.312	0.134	0.158	0.081
Size range (g)	12-191	41-113	45-164	53-56
S				
n	20	8	14 ^b	13 ^b
$\bar{X}$	0.277	1.697	1.054	0.771
S.D.	0.121	0.718	0.312	0.270
Size range (g)	1-47	2-4	1-22	1-8

^aRB = rock bass (Ambloplites rupestris), HS = hog sucker (Hypentelium nigricans), S = shiners (Nortropis spp.).

^bFor the smaller sized fish, the whole fish was analyzed for total mercury.

Table 14. Total mercury concentration in axial muscle (except where noted) of 3 fish species collected in the North Fork of the Holston River, August 1975. All values are $\mu\text{g/g}$ wet wt

Species ^a	Chatham Hill 43 km above plant	Neil Bridge 9 km above plant	Downstream disposal ponds 3.7 km below plant	McKenna Island 9.7 km below plant	Hayters Gap 21 km below plant	Hines Island 43 km below plant	Kingsport 133 km below plant
RB							
n	10	10	NC ^b	10	NC ^b	10	10
X	0.219	0.322	-	1.814	-	1.292	1.122
S.D.	0.128	0.143	-	0.497	-	0.298	0.899
Size range (g)	12-101	9-134	-	24-92	-	6-165	42-113
HS							
n	10	10	10	10	10	10	10
X	0.194	0.278	2.140	2.190	1.661	1.110	0.819
S.D.	0.089	0.212	0.933	0.465	0.334	0.265	0.570
Size range (g)	15-283	15-274	36-311	26-536	37-250	18-471	49-385
S							
n	5	5	5 ^c	5 ^c	5	4	5
X	0.129	0.121	2.122	2.340	1.344	1.430	0.515
S.D.	0.044	0.031	0.809	0.152	0.570	0.528	0.575
Size range (g)	7-10	8-12	2-3	2-3	9-11	8-13	8-12

^aRB = rock bass (*Ambloplites rupestris*), HS = hog sucker (*Hypentelium nigricans*), S = shiners (*Nortropis* spp.).

^bNC = No fish collected.

^cFor the smaller sized fish, the whole fish was analyzed for total mercury.

Table 15. Percentage methylmercury in axial muscle for fish species in the North Fork Holston River, August 1974 and August 1975

	Chatham Hill 43 km above plant 1975	McCrary 5.5 km above plant 1974	Downstream disposal ponds 3.7 km below plant 1974	McKenna Island 9.7 km below plant 1975	Hayters Gap 21 km below plant 1974	Kingsport 133 km below plant 1974
Rock bass (<u>Amblopytes rupestris</u>)						
n	10	10	7	10	6	3
$\bar{X}$ % MeHg	94.5	86.7	83.7	86.4	99.5	78.9
$\pm$ 2 S.E.	$\pm$ 3.4	$\pm$ 8.3	$\pm$ 7.0	$\pm$ 5.6	$\pm$ 4.0	$\pm$ 16.9
Hog sucker (<u>Hypentelium nigricans</u>)						
n	10	9	3	10	6	2
$\bar{X}$ % MeHg	96.0	103.8	90.9	94.0	86.3	84.6
$\pm$ 2 S.E.	$\pm$ 2.7	$\pm$ 3.3	$\pm$ 13.8	$\pm$ 4.5	$\pm$ 5.8	$\pm$ 28.7
Shiners (<u>Notropis</u> spp.)						
n	5	10	3	4	5	6
$\bar{X}$ % MeHg	92.9	84.5	105.1	84.6	83.8	78.2
$\pm$ 2 S.E.	$\pm$ 2.4	$\pm$ 11.2	$\pm$ 5.2	$\pm$ 2.26	$\pm$ 10.7	$\pm$ 7.3

at Kingsport, 1974) to a high of 105.1% (shiners below the disposal ponds, 1974). The mean percentage methylmercury of all 119 fish analyzed was $90.26\% \pm 2.08\%$ ($\bar{X} \pm 2$ S.E.).

Significant regressions ($P = 0.05$) of total mercury concentration in rock bass and hog sucker on fish weight were observed for some but not all of the possible station-date combinations in the North Fork Holston River. Significant regressions of mercury concentration in rock bass on weight were observed for seven of the nine possible station-date combinations. For hog sucker, significant regressions of 5 out of 11 station-date pairs were observed. The proportion of total variance in mercury concentration explained by weight (r^2) for rock bass ranged from 0.427 to 0.952. For hog suckers, r^2 values ranged from 0.414 to 0.870 (Table 16).

The results of total mercury and methylmercury analyses of benthic invertebrates collected in the North Fork Holston River in 1974 and 1975 are given in Tables 17 and 18 respectively. In both 1974 and 1975, ΣHg concentration in benthic invertebrates was lowest above the chloralkali plant, reached a maximum immediately below the plant, then decreased downstream. Total Hg concentrations in individual taxonomic groups ranged from a high of $3.75 \mu\text{g/g}$ (Hydropsychidae, 3.7 km below plant, 1975) to $0.016 \mu\text{g/g}$ (Psephenidae, 5.5 km above the plant, 1974). In both 1974 and 1975, mean ΣHg levels of all taxonomic groups analyzed 133 km below the plant were higher than mean ΣHg levels above the plant.

The percentage MeHg of ΣHg observed for individual taxonomic groups from all stations ranged from a low of 29% (Hydropsychidae,

Table 16. Summary statistics for significant regressions of total mercury concentration in axial muscle Y ($\mu\text{g/g}$ wet wt) as a function of fish weight X (g wet wt) for rock bass and hog sucker, North Fork Holston River^{a,b,c}

Species ^d	Station (year sampled)	Equation	n	r ²
RB	McCrady (1974)	$Y = 0.0031X + 0.338$	13	0.501
RB	Below ponds (1974)	$Y = 0.00854X + 1.196$	11	0.582
RB	Hayters (1974)	$Y = 0.1382X + 0.932$	16	0.608
RB	Kingsport (1974)	$Y = 0.01312X$	10	0.952
RB	Chatham (1975)	$Y = 0.00427X$	10	0.941
RB	Hines (1975)	$Y = 0.0039X + 1.095$	10	0.427
RB	Kingsport (1975)	$Y = 0.01843X$	10	0.785
HS	McCrady (1974)	$Y = 0.00409X$	11	0.689
HS	Chatham (1975)	$Y = 0.00492X + 0.107$	10	0.830
HS	Neil (1975)	$Y = 0.00223X$	10	0.870
HS	McKenna (1975)	$Y = 0.00191X + 1.875$	10	0.414
HS	Kingsport (1975)	$Y = 0.0031X$	10	0.841

^aSignificance level for regression coefficient and intercept is $P \leq 0.05$. If an intercept was not significant, the regression was computed again with no intercept.

^bRegression for rock bass at stations 2 and 5, 1975, were not significant. Regression for hog sucker at stations 4, 6, and 8, 1974, and 4, 6, and 7, 1975, were not significant.

^cSee Table 7 for station identification.

^dRB = rock bass (*Ambloplites rupestris*), HS = hog sucker (*Hypentelium nigricans*).

Table 17. Total mercury (Σ Hg) and percentage methylmercury (MeHg) in benthic invertebrates from the North Fork Holston River, August 1974. All values are μ g/g wet wt^a

Taxa	McCrary 5.5 km above plant		Below ponds 3.7 km below plant		Hayters Gap 21.0 km below plant		Kingsport 133 km below plant	
	Hg (μ g/g)	MeHg (%)	Hg (μ g/g)	MeHg (%)	Hg (μ g/g)	MeHg (%)	Hg (μ g/g)	MeHg (%)
Trichoptera Hydropsychidae	b	b	1.680 (1)	29 (1)	0.560 (1)	41 (1)	b	b
Megaloptera <u>Corydalus</u>	0.088 (1)	66 (1)	1.940 (1)	41 (1)	0.640 (3)	49 (3)	0.222 (4)	64 (3)
Decapoda	0.065 (2)	52 (2)	b	b	1.120 (1)	48 (1)	0.270 (2)	49 (2)
Coleoptera Psephenidae	0.016	37	b	b	b	b	b	b
Combined remaining taxa	0.035 (1)	51 (1)	1.030 (1)	83 (1)	0.840 (1)	40	0.110 (1)	a (1)
Mean of taxa analyzed $\pm$ 2SE	0.051 $\pm$ 0.032 (4)	51 $\pm$ 12 (4)	1.550 $\pm$ 0.552 (3)	51.1 $\pm$ 33.7 (3)	0.790 $\pm$ 0.249 (4)	44 $\pm$ 5 (4)	0.201 $\pm$ 0.095 (3)	56 $\pm$ 15 (2)

^aSee Table 7 for station identification. Numbers in parentheses indicate number of replicates.

^bTaxa not collected in sufficient quantity to analyze.

Table 18. Total mercury (ΣHg) and percentage methylmercury (MeHg) in benthic invertebrates from the North Fork Holston River, August 1975. All values are in $\mu\text{g/g}$ wet wt^a

Taxa	Chatham Hill 43.0 km above plant		Neil Br. 9.0 km above plant		Below ponds 3.7 km below plant		McKenna 9.7 km below plant		Hayters Gap 21.0 km below plant		Hines 43.0 km below plant		Kingsport 133 km below plant	
	Hg		Hg		Hg		Hg		Hg		Hg		Hg	
	($\mu\text{g/g}$)	(%)	($\mu\text{g/g}$)	($\mu\text{g/g}$)	($\mu\text{g/g}$)	(%)	($\mu\text{g/g}$)	(%)	($\mu\text{g/g}$)	($\mu\text{g/g}$)	($\mu\text{g/g}$)	($\mu\text{g/g}$)	($\mu\text{g/g}$)	($\mu\text{g/g}$)
Trichoptera Hydropsychidae	0.037 (1)	57 (1)	0.035 (1)	3.750 (2)	0.745 (2)	54 (2)	0.560 (2)		0.370 (2)		0.215 (2)			
Megaloptera Corydalus	0.054 (10)	63 (10)	0.062 (2)	1.620 (2)	1.054 (10)	63 (10)	0.695 (2)		0.440 (2)		0.220 (2)			
Diptera Atherix	0.081 (2)	44 (2)	0.050 (2)	1.430 (2)	0.830 (2)	77 (2)	0.590 (2)		0.440 (2)		0.215 (2)			
Decapoda	0.038 (2)	80 (2)	0.027 (2)	1.315 (2)	b		0.380 (2)		b		b			
Mean of taxa analyzed $\pm$ 2SE	0.052 $\pm$ 0.020 (4)	61 15 (4)	0.043 $\pm$ 0.016 (4)	2.080 $\pm$ 1.154 (4)	0.876 $\pm$ 0.219 (3)	65 13 (3)	0.556 $\pm$ 0.131 (4)		0.416 $\pm$ 0.046 (3)		0.217 $\pm$ 0.005 (3)			

^aSee Table 7 for station identification. Numbers in parenthesis indicate number of replicates.

^bTaxa not collected in sufficient quantity to analyze.

3.7 km below plant, 1974) to 83% (combined remaining taxa, 3.7 km below plant, 1974). The mean percentage MeHg for all taxonomic groups analyzed at a given station ranged from 44% (21 km below plant, 1974) to 65% (9.7 km below plant, 1975) (Tables 17 and 18).

The data used in the multiple regression analysis of factors affecting the mercury concentration in fish and benthic invertebrates in the North Fork are given in Table 19. In the multiple regression analysis for fish species, the dependent variables used were mercury concentration in rock bass or hog sucker. Independent variables included total mercury concentration in the water column (particulate and dissolved), total mercury concentration dissolved in the water column, total mercury concentration in bed sediment (< 44- μ m size), and total mercury concentration in benthic invertebrates. In the "backward elimination" procedure we employed, the regression is first calculated with all independent variables included. The variable showing the smallest contribution to the model (on the basis of a "partial F statistic") is then deleted and the regression computed on the remaining variables. This process was repeated until only one variable remained in the model. The r^2 values for the regression analysis of rock bass indicated 96.4% of the variability in mercury concentration was accounted for by the four independent variables. Two variables (Hg concentration in benthos and Hg concentration in sediment) were contributing significantly to the regression at the $P \leq 0.01$ level. Mercury concentration in the water column and mercury concentration dissolved in water were not contributing significantly to the regression at the $P \leq 0.05$ level. The r^2 values for the regression

Table 19. Summary of data used for "backward elimination" multiple regression analysis of factors affecting total mercury concentration in rock bass, hog sucker, and invertebrates, North Fork Holston River. All values are $\mu\text{g/g}$. Sediment values are dry weight; fish and invertebrate values are wet weight

Year	Station ^a	[Total Hg] ^b in water	[Total Hg] ^b dissolved in water	[Total Hg] ^b in bed sediment ($< 44 \mu\text{m}$)	[Total Hg] ^c in benthic invertebrates	[Total Hg] in rock bass	[Total Hg] in hog sucker
1974	McCrary	0.077	0.033	0.340	0.051	0.493 ^d	0.409 ^d
1974	Below ponds	0.220	0.186	20.200	1.550	1.623 ^d	1.420 ^e
1974	Hayters	0.100	0.062	6.300	0.790	1.623 ^d	1.235 ^e
1974	Kingsport	0.048	0.008	1.700	0.201	0.656 ^d	0.427 ^e
1975	Below ponds	0.210	0.056	18.500	2.028	f	2.140 ^e
1975	Hayters	0.067	0.027	4.300	0.556	f	1.661 ^e
1975	Kingsport	0.025	0.018	0.920	0.217	0.921 ^d	0.371 ^d
1975	Chatham	0.070	0.001	0.140	0.052	0.213 ^d	0.198 ^d
1975	Neil	0.034	0.000	0.160	0.043	0.322 ^d	0.223 ^d
1975	McKenna	0.074	0.040	9.200	0.876	1.814 ^d	2.066 ^d
1975	Hines	0.032	0.021	2.400	0.416	1.290 ^d	1.110 ^e

^aSee Table 7 for station identification.

^bData obtained from Fulkerson et al. (1974) and Van Hook and Shultz (1976).

^cData from Tables 17 and 18.

^dConcentration for 50-g rock bass or 100-g hog sucker from regression equations (Table 16).

^eConcentration is mean for all fish analyzed at station and date, since regression equations were not significant (Table 13 and 14).

^fRock bass were not collected.

analysis of hog sucker indicated 80.5% of the variability of mercury concentration was accounted for by the four independent variables. Two variables (Hg concentration in benthos and Hg concentration in the water column) contributed significantly to the regression at the $P \leq 0.05$ level (Table 20). Mercury concentration in sediment and mercury concentration dissolved in water did not contribute significantly to the regression.

The multiple regression analysis utilizing Hg concentration in benthic invertebrates as the dependent variable, and Hg concentration in the water column, dissolved in water and Hg concentration in sediment as the independent variables indicated 97.4% of the variance in mercury concentration in invertebrates was accounted for by the independent variables (Table 21). Concentration of total mercury in sediment alone accounted for 94.1% of the variance. Addition of total mercury concentration dissolved in water accounted for 97.4% of the variance, and both variables were significant at $P < 0.052$ level. Adding total mercury concentration in the water column did not increase the percentage of variance accounted for.

From the standpoint of environmental monitoring for mercury contamination, it would be very useful to be able to predict mercury levels in biotic compartments from an easily monitored parameter in the physical environment. We examined this possibility by calculating the regressions of total mercury concentration in rock bass, hog sucker, and benthic invertebrates on sediment mercury concentration for the station-data combinations available (Table 19). The relation of total mercury concentration in sediment to total mercury concentration in

Table 20. Summary of factors affecting total mercury concentrations in rock bass and hog sucker, North Fork Holston River, utilizing "backward elimination" multiple regression analysis

Dependent variable	Independent variables in model ^a	r ²
Hg concentration in Rock bass	1,2,3,4	0.964
	1,3,4	0.964
	3,4	0.937
	4	0.722
Hg concentration in Hog sucker	1,2,3,4	0.805
	1,3,4	0.805
	1,4	0.803
	4	0.677

^a1 = Hg concentration water column, 2 = Hg concentration dissolved in water, 3 = Hg concentration in sediment (< 44- μ m size), and 4 = Hg concentration in benthic invertebrates.

Table 21. Summary of factors affecting total mercury concentration in benthic invertebrates, North Fork Holston River, utilizing "backward elimination" multiple regression analysis

Dependent variable	Independent variables in model ^a	r ²
Hg concentration in benthic invertebrates	1 2 3	0.974
	2 3	0.974
	3	0.941

^a1 = Hg concentration water column, 2 = Hg concentration dissolved in water, and 3 = Hg concentration in sediment (< 44- μ m size).

rock bass and hog sucker is given in Fig. 16. For both species, the relationship appeared to be logarithmic. We therefore calculated a linear regression of $\ln(\text{Hg concentration in sediment} + 1)$ on total mercury concentration in fish (rock bass and hog sucker pooled). The regression results are given in Fig. 16. Solving the regression for fish concentration equal to $0.5 \mu\text{g/g}$ results in a sediment concentration of $0.944 \mu\text{g/g}$. This analysis indicates that when sediment ($< 44 \mu\text{m}$) Hg levels exceed approximately $1 \mu\text{g/g}$, rock bass weighing 50 g or hog sucker weighing 100 g may contain mercury near the FDA "action level" in the North Fork.

The relation of total mercury concentration in benthos to total mercury concentration in sediment ($< 44 \mu\text{m}$) is shown in Fig. 17 with the calculated regression. The relation is linear throughout the entire range of sediment concentration. This finding indicates that mercury concentration in benthic invertebrates appears to reflect mercury concentration in sediment quite precisely.

Mercury Concentration in Fish Species Collected in Cherokee Reservoir

The results of total mercury analyses for fish species collected in Cherokee Reservoir in 1974 and 1975 are given in Table 22. With the single exception of smallmouth bass collected below the steam plant in 1974 ($\bar{X} = 0.504 \mu\text{g/g}$), all mean total mercury concentrations for the other species in Cherokee Reservoir at each station in 1974 and 1975 were below the FDA "Action Level" of $0.5 \mu\text{g/g}$. A total of 155 largemouth bass, bluegill, and smallmouth bass were analyzed for total mercury from Cherokee Reservoir. Total mercury in ten of these

Fig. 16. Relation of total mercury concentration in sediment to mercury concentration in rock bass and hog sucker. Data from Table 19.

Fig. 17. Relation of mean total mercury concentration in benthic invertebrates to total mercury concentration in sediment. Data from Table 19.

Table 22. Total mercury concentration in axial muscle (except where noted) of fish species collected in Cherokee Reservoir, August 1974 and 1975 (all values are $\mu\text{g/g}$ wet wt)

	Below Sevier detention pond 188 km below plant 1974	25E bridge 238 km below plant 1974	Cherokee Dam 277 km below plant 1974	Below Sevier detention pond 188 km below plant 1975	Cherokee Dam 277 km below plant 1975
Largemouth bass (<i>Micropterus salmoides</i>)					
n	12	19	15	10	10
$\bar{X}$	0.345	0.093	0.190	0.204	0.114
SD	0.277	0.053	0.086	0.127	0.149
Size Range (gn)	130-2073	19-726	58-451	5-2721	24-1020
Bluegill (<i>Lepomis macrochirus</i>)					
n	20	20	19	9	10
$\bar{X}$	0.325	0.127	0.155	0.212	0.100
SD	0.149	0.073	0.085	0.165	0.036
Size Range (gn)	35-156	11-74	7-103	5-109	12-139
Gizzard shad (<i>Dorosoma cepedianum</i>)					
n	20	20	20	10	10
$\bar{X}$	0.089	0.059	0.055	0.062	0.053
SD	0.036	0.021	0.011	0.046	0.012
Size Range (gn)	2-284	2-134	18-174	25-54	134-250
Threadfin shad (<i>Dorosoma petenense</i>)					
n	20 ^a	20 ^a	20 ^a	NAB ^b	NAB ^b
$\bar{X}$	0.048	0.029	0.047	-	-
SD	0.012	0.018	0.028	-	-
Size Range (gn)	1-4	1-15	1-49	-	-
Smallmouth bass (<i>Micropterus dolomieu</i>)					
n	7	3	NCC ^c	NCC ^c	NCC ^c
$\bar{X}$	0.504	0.111	-	-	-
SD	0.515	0.051	-	-	-
Size Range (gm)	35-1477	6-208	-	-	-

^aFor the smaller sized fish, the whole fish was analyzed for total mercury.

^bNA = No fish analyzed.

^cNC = No fish collected.

individual fish analyzed was above 0.5 $\mu\text{g/g}$. Nine of these fish above 0.5 $\mu\text{g/g}$ total mercury were collected at the upper end of the reservoir. None of the 140 shad analyzed for total mercury were above 0.5 $\mu\text{g/g}$.

The relationship between total mercury concentration and weight was examined for each species at each station on each collection date (Table 23). Significant regressions ($P < 0.05$) were observed for 9 of the 20 species-station-date collections. The percentage of variance in total mercury concentration explained by weight in those collections where significant regressions were observed ranged from 74.6 to 96.8% (Table 23).

The percentage methylmercury of total mercury in fish analyzed ranged from 54.3% for threadfin shad collected at the Rt. 25E bridge in 1974 to 93.3% for bluegill collected below the steam plant in 1975 (Table 24). The mean percentage methylmercury of all 156 fish analyzed was $75.06 \pm 2.96\%$ ($\bar{X} \pm 2 \text{ S.E.}$). For largemouth bass and bluegill, it appears that percentage methylmercury is generally lower at the stations in the middle and lower areas of Cherokee Reservoir, although our data are not extensive enough to explore this trend in any depth.

Mercury concentrations in the Cherokee striped bass were high relative to the FDA "Action Level," with average values in muscle tissue for each collection near or above the 0.5- $\mu\text{g/g}$ limit (Table 25). The five fish collected in June 1972 were especially high in mercury, averaging $2.29 \pm 0.38 \mu\text{g/g}$ (1 S.E.) in muscle tissue. This high level of mercury in dead and moribund fish led to speculation that Hg poisoning might have been responsible. However, the sample size (5) was regarded as too small for a definitive conclusion. The nine

Table 23. Statistics for significant ($P < 0.05$) regressions of total mercury concentration ($\mu\text{g/g}$ wet wt) on fish weight (g wet wt) for fish collected in Cherokee Reservoir, 1974 and 1975^{a,b}

Species	Station and (date)	Equation	n	P	r ²
Largemouth bass	Below Steam Plant (1974)	$Y = 0.00048X + 0.112$	12	0.0001	0.968
	25E Bridge (1974)	$Y = 0.00025X + 0.0569$	19	0.0001	0.726
	Dam (1974)	$Y = 0.00073X + 0.0484$	16	0.0001	0.806
	Dam (1975)	$Y = 0.00049X + 0.0211$	10	0.0001	0.967
Bluegill	25E bridge (1974)	$Y = 0.00309X$	20	0.0001	0.889
	Dam (1975)	$Y = 0.00075X + 0.0584$	10	0.0007	0.777
Smallmouth bass	Below steam plant (1974)	$Y = 0.00093X + 0.0595$	7	0.0009	0.770
Threadfin shad	25E Bridge (1974)	$Y = 0.00533X + 0.0181$	20	0.0001	0.885
	Dam (1974)	$Y = 0.00202X + 0.0297$	20	0.0001	0.746

^aSee Table 7 and Figure 8 for station location.

^bIf intercepts of regressions were not significant ($P < 0.05$), regressions were computed through the origin.

Table 24. Percentage methylmercury in fish species collected from Cherokee Reservoir, August 1974 and 1975

	Below Sevier detention pond 188 km below plant 1974	Below Sevier detention pond 188 km below plant 1975	25E bridge 238 km below plant 1974	Cherokee Dam 277 km below plant 1974
Largemouth bass (<u>Micropterus salmoides</u>)				
n	10	10	10	10
X% MeHg	92.5	93.1	77.4	68.5
± 2 S.E.	6.2	3.8	7.9	8.7
Bluegill (<u>Lepomis macrochirus</u>)				
n	11	9	10	9
X% MeHg	82.8	93.3	73.6	65.9
± 2 S.E.	13.8	2.3	11.7	8.4
Gizzard shad (<u>Dorosoma cepedianum</u>)				
n	11	10	10	10
X% MeHg	77.3	70.9	61.8	70.1
± 2 S.E.	10.9	9.2	14.6	8.3
Threadfin shad (<u>Dorosoma petenense</u>)				
n	10	NC ^a	9	7
X% MeHg	57.9	-	54.3	81.8
± 2 S.E.	6.8	-	13.6	5.9
Smallmouth bass (<u>Micropterus dolomieu</u>)				
n	7	NC ^a	3	NC ^a
X% MeHg	80.0	-	79.9	-
± 2 S.E.	10.6	-	16.1	-

^aNC = No fish collected.

Table 25. Total mercury (ΣHg) concentration ($\mu\text{g/g}$ wet wt) and percentage methylmercury (MeHg) in striped bass (Morone saxatilis) from Cherokee Reservoir, 1972-1974^a

Date collected	n	Whole fish size range (kg)	Organ	Hg ($\bar{X} \pm 1 \text{ S.E.}$)	% MeHg ($\bar{X} \pm 1 \text{ S.E.}$)
6/72	5	5-6	Muscle	2.29 $\pm$ 0.38	a
1/73	9	1-6	Muscle	0.47 $\pm$ 0.15	a
1/73	9		Liver	0.321 $\pm$ 0.037	a
1/73	9		Brain	0.706 $\pm$ 0.139	a
1/73	9		Gonads	0.102 $\pm$ 0.022	a
4/73	11	2-8	Muscle	0.42 $\pm$ 0.12	76 $\pm$ 7
4/73	11		Liver	0.385 $\pm$ 0.093	55 $\pm$ 7
4/73	11		Brain	0.027 $\pm$ 0.004	37 $\pm$ 6
4/73	11		Gonads	0.071 $\pm$ 0.013	85 $\pm$ 5
6-8/73	13	2-6	Muscle	0.54 $\pm$ 0.02	a
6-8/73	13		Liver	0.55 $\pm$ 0.06	a
8/74	10	2-7	Muscle	0.51 $\pm$ 0.051	50 $\pm$ 3

^aMeHg analysis not performed.

striped bass collected in January 1973 had lower muscle mercury concentrations than the June 1972 fish [$0.47 \pm 0.15 \mu\text{g/g}$ (1 S.E.)]. Liver, brain, and gonad tissues were analysed in the striped bass collected in January 1973. The highest tissue level of mercury was brain with $0.70 \pm 0.139 \mu\text{g/g}$.

In the 11 striped bass taken in April 1973, mercury concentrations were similar to the January 1973 collection with the exception of brain, which was much lower in the April fish. In addition to ΣHg , methylmercury (MeHg) was also analysed, and as is usually the case, most of the ΣHg ($76 \pm 7\%$) was in the MeHg form in muscle tissue.

In 1973, 13 striped bass were collected in June, July, and August by the Tennessee Game and Fish Commission in the lower end of Cherokee Reservoir. The ΣHg concentrations were not significantly different ($P < 0.05$) in muscle and liver. Muscle Hg concentrations were somewhat higher than in April 1973, but still much lower than in 1972.

The following year (August 1974), 10 striped bass were collected at Mossy Creek embayment (near Cherokee Dam) by hook and line. Muscle tissue only was analysed for ΣHg and MeHg. Although the ΣHg concentrations in 1974 were very similar to the concentrations for the summer 1973 collection, the MeHg measurements were much lower than those in April 1973.

Discussion

The results of our analyses of total mercury in fish species in the North Fork Holston River clearly indicate accumulation of mercury in fish muscle tissue to above or near the $0.5\text{-}\mu\text{g/g}$ FDA "Action Level"

in effect at that time for at least 133 km downstream from the chloralkali plant. In 12 out of 22 station-date-species data sets in the North Fork, we observed a significant regression of total mercury concentration on fish weight (Table 16). Size and age of fish have been previously shown to influence levels of mercury accumulation in fish (Richins and Risser 1975, Annett et al. 1975, Potter et al. 1975, Koirttyohann et al. 1974, D'Itri et al. 1971, Mathis and Kevern 1975, Kelly et al. 1975, Scott and Armstrong 1972, Scott 1974).

Since size and age of fish are not independent, it is difficult to determine the relative importance of duration of exposure and increase in mass of tissue in mercury accumulation during the exposure period. Intuitively, we would have expected to observe significant relationships of total mercury concentration on fish weight for all of our collections, if indeed weight adequately reflected age (which would reflect duration of exposure). Several possible explanations can be offered for collections where a relationship was not observed. Inadequate sample size and/or inadequate representation of size classes in our samples could have masked any relationship. In addition, since our sampling stations in the North Fork were in a continuum above and below the mercury source with no barriers to fish movement, the fish collected at any given station could have experienced different mercury exposure conditions at different stages in their life history.

Our observation that $90.26 \pm 2.08\%$ ($\bar{X} \pm 2$ S.E.) of the total mercury in 119 fish analyzed in the North Fork was in the methylated form is consistent with the majority of published information (Noren and Westoo 1967, Zitko et al. 1971, Kamps et al. 1972, Lockhart et al.

1972, Westoo 1973, Bache et al. 1971, Bishop and Neary 1974, Huckabee et al. 1974). We did not observe any apparent consistent difference in percentage methylmercury in fish species above and below the chloralkali plant (Table 15), which supports the observation of Zitko (1971). We did not examine the relationship of percentage methylmercury and fish size due to inadequate sample size. Bache et al. (1971) observed an increasing proportion of methylmercury to total mercury in whole lake trout with age over the first three years of life. In older fish, the percentage methylmercury was independent of age. Westoo (1973) and Bishop and Neary (1974) did not observe any significant relationship of percentage methylmercury in fish muscle and age.

Our examination of total mercury concentration in whole benthic invertebrates (Tables 17 and 18) indicates invertebrate taxa in the North Fork above and below the chloralkali plant accumulate mercury in a pattern similar to that observed in fish species. Total mercury concentration in benthic invertebrates above the chloralkali plant was in the 0.05- $\mu\text{g/g}$ range, which is consistent with observation of total mercury concentration in freshwater invertebrates from a wide spectrum of "uncontaminated" environments (Johnels et al. 1967, Armstrong and Hamilton 1973, Huckabee and Hildebrand 1974, Hasselrot and Gothberg 1974, Cox et al. 1975). Total mercury concentrations in benthic invertebrate taxa immediately below the chloralkali plant fell between 1 and 4 $\mu\text{g/g}$ (Tables 17 and 18). Total mercury concentrations in benthic invertebrates in this range or greater have been observed in several "contaminated environments" (Johnels et al. 1967, Armstrong and Hamilton 1973, Cox et al. 1975).

Our observation that a significant fraction (29 to 83%, Tables 17 and 18) of total mercury concentration in benthic invertebrates, both above and below the plant, is in the methylated form is contradictory to results published by Jernelov and Lann (1971) (5 - 20%) and Cox et al. (1975) (< 1 - 17%), but in line with observations of Huckabee and Hildebrand (1974) (23 to 100%) and Trudel et al. (1977) (0 to 96%).

Our estimates of mercury concentration in benthic invertebrates from the North Fork Holston River, as well as other estimates cited, are for the whole animal including the gastrointestinal tract. These estimates are potentially biased by mercury contamination on the surface of the organisms and possible sediment contamination in the gastrointestinal systems. Elwood et al. (1976) examined the role of gut contamination in whole body concentration of 30 elements in Tipula spp. from an uncontaminated stream. For 20 elements the whole animal concentration after gut elimination was significantly less than before gut elimination. The difference in total mercury contamination before and after gut evacuation in Tipula was not significant (Elwood et al. 1976). In one species from an uncontaminated environment, it appears gut contents are not a significant source of mercury contamination. Surface and gut contamination, however, remains as a factor explaining the discrepancies in reported values for percentage methylmercury in benthic invertebrates.

The possibility of surface or gut contamination in estimates of mercury concentration in benthic invertebrates is important if research is directed toward kinetics of accumulation and excretion of mercury by invertebrates. If research is directed toward examination of trophic

transfer of mercury through food chains, the concentration of mercury in the entire animal is the relevant parameter since predators consume the entire animal, surface and gut contamination included. The total mercury levels and percentage methylmercury in benthic invertebrates in the North Fork Holston River indicate that in this system, benthic invertebrates can be a significant source of methylmercury for rock bass and hog sucker. The diets of both species include benthic invertebrates (Carlander 1969, 1977).

We were mildly surprised at the large percentage of variance in mercury concentration in rock bass and hog sucker explained by the independent variables included in our multiple regression analysis (Table 20). Considering the inherent variability encountered in aquatic ecosystems, we think these results are meaningful. We emphasize that we are not implying strict cause-and-effect mechanisms in this analysis. Our results demonstrate that abiotic factors can be related to observed total mercury levels in fish species, if a reasonably comprehensive data set is collected. The amount of effort devoted to the biotic and abiotic sampling in the North Fork is not always feasible. Our attempt to relate mercury concentration in rock bass and hog sucker to a single variable in the environment (Hg concentration in $< 44\text{-}\mu\text{m}$ sediment) is encouraging. We do not infer from this observed relationship that fish are obtaining mercury from the sediment. We infer that sediment mercury levels may be an accurate reflection of overall mercury exposure conditions. The explanation of why the relationship of concentration in sediment to fish appears logarithmic, is not intuitively obvious. The logarithmic relationship

may indicate that elimination rate of mercury by fish at higher tissue concentrations is retarded.

The multiple regression analysis of factors affecting mercury concentration in benthic invertebrates again indicates a very high percentage of variance in benthos mercury concentration which may be explained by mercury concentration in water and sediment (Table 21). Sediment concentration alone accounted for 94% of the variance. The linear regression of mercury concentration in benthos on mercury concentration in sediment may indicate that surface and/or gut contamination is significant. If mercury concentration in benthos is expressed on a dry weight basis using a ratio of dry weight/wet weight equal to 0.1 (Winberg 1971), benthic invertebrates appear to have about the same mercury concentration as sediment (Fig. 12). The role of such contamination in benthic invertebrate mercury analyses should be investigated.

Mercury concentration in fish species collected in Cherokee Reservoir are generally below the 0.5- $\mu\text{g/g}$ FDA "Action Level" (Table 22) and the majority of mercury in muscle tissue is in the methylated form (Table 24). Predatory fish (bass, bluegill) consistently have higher mean mercury concentrations than both threadfin shad and gizzard shad (Table 22). As was the case in the North Fork, we observed significant regressions of total mercury concentration on weight for some but not for all species-station-data analyses in Cherokee Reservoir (Table 23). The reasons postulated for this discrepancy given in the North Fork discussion most likely apply to Cherokee Reservoir as well. Ten (6.4%) of the 155 bass and bluegill collected from Cherokee Reservoir were

above the "Action Level" in effect at that time. Nine of these fish above 0.5 $\mu\text{g/g}$ were collected from the upper reservoir, indicating qualitatively that mercury levels in these species are higher nearer the confluence of the Holston River with the reservoir. Of the 10 individual fish above the FDA "action level," eight had muscle mercury concentrations between 0.5 and 0.6 $\mu\text{g/g}$, and two were between 1.0 and 2.0 $\mu\text{g/g}$. Of the 128 rock bass and hog sucker collected in the North Fork Holston River below the chloralkali plant, 120 (93.7%) were above 0.5 $\mu\text{g/g}$ total mercury in axial muscle. Twenty-six of these fish were above 2.0 $\mu\text{g/g}$ and 68 were above 1.0 $\mu\text{g/g}$.

The observation that mercury levels in fish in the North Fork above the chloralkali plant in some cases appeared elevated (Table 13) deserves note. Nine of 31 rock bass and 1 of 31 hog suckers collected above the plant had mercury levels above 0.5 $\mu\text{g/g}$. The explanation for these elevated values is not immediately clear but could be caused by both diffuse sources of mercury to the environment and migration of fish in the near plant area causing variable mercury exposure. The majority of fish (rock bass and hog suckers) with elevated levels above the plant were at the station 5.5 km above the plant.

The analysis of mercury concentration in axial muscle of striped bass from Cherokee Reservoir in the period 1972 to 1974 seems to indicate levels declined after 1972 (Table 25). However, the 1972 fish analyzed were dead or moribund at the time of capture, and whether this affected mercury analyses is not known. In the January 1973 collection of striped bass (Table 25), higher levels were observed in the brain and muscle than in the liver. This observation may indicate that the

striped bass in Cherokee Reservoir are exposed to mercury episodically, because in fish, mercury accumulates rapidly in the liver, and then is slowly redistributed to and accumulates in muscle and brain (Massaro and Giblin 1972). Therefore, fish under constant mercury exposure tend to have higher concentrations of mercury in liver than in muscle and brain. The total mercury concentration in muscle of striped bass collected in 1974 was similar to that in fish collected in 1973, but the percentage of methylmercury in muscle was lower in 1974. Because methylmercury is known to have a long retention time in fish, it tends to slowly accumulate for the duration of the fish life span (Reichert and Malins 1974). Inorganic mercury, on the other hand, while absorbed almost as readily as methylmercury, is depurated much faster than methylmercury (Reichert and Malins 1974). Methylmercury concentration is therefore a better gauge of the fishes' exposure history to mercury contamination. If the decrease in methylmercury in muscle in 1974 is real, this suggests that these fish may have been exposed to less total mercury than the April 1973 fish. Episodic inputs of total mercury, while diminishing in quantity over time, could temporarily raise muscle total mercury concentration following the event, while methylmercury uptake rate may be increased to a lesser degree.

The potential explanation for the die-offs of striped bass in Cherokee Reservoir have been discussed by Coutant (1978). He hypothesizes that the die-offs may result from concurrent summertime oxygen depletion and increased temperatures of the reservoir. These conditions may stress the striped bass to the point where susceptibility to metal toxicity is increased.

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5. SUMMARY OF MAJOR OBSERVATIONS

This chapter briefly summarizes the observations presented in Chapters 2 to 4 of this report. The reader is cautioned to refer to the individual chapters for detailed explanations on qualifications to these summary statements.

Chemistry and Emissions of Mercury From Chloralkali Solid Waste Ponds

1. The major component of the solid in the chloralkali waste ponds was calcium carbonate. Other major ions include sodium, magnesium, aluminum, iron, and silica (present as oxides, hydroxides, carbonates, sulfates, or silicates).
2. Leachate from the two waste ponds was a caustic brine composed primarily of chlorides and hydroxides of sodium and calcium.
3. The total mercury concentration in the solid waste in the two disposal ponds ranged from 4.4 to 350 $\mu\text{g/g}$ ($\bar{x}$ = 92 $\mu\text{g/g}$), with higher concentrations tending to occur near the surface and in pond #1.
4. The total mercury concentration in leachate from the waste ponds ranged from 11 to 230 $\mu\text{g/liter}$ ($\bar{x}$ = 68 $\mu\text{g/liter}$), with pond 1 leachates exhibiting considerably higher concentrations than pond 2 leachates. The mercury in leachates was predicted to exist as hydroxy and chloro complexes, as well as dissolved elemental Hg.
5. Relatively little of the total mercury in the solid waste ponds was leachable under laboratory experimental conditions (< 5%).

6. The atmospheric emission of elemental mercury vapor from the waste material was related to surface temperature and was estimated to be 120 to 170 $\mu\text{g}/\text{m}^2/\text{h}$ in laboratory experiments.
7. Measured and predicted air concentrations of mercury vapor indicated that the entire surface area of the waste ponds emitted mercury vapor at a rate comparable to laboratory estimates.
8. During warm months, the concentration of mercury in air measured near the waste ponds approached the EPA ambient air guideline of 1000 ng/m^3 .
9. Nearly 100% of the mercury in air over the waste ponds was in the vapor form, with very little additional particulate Hg present.
10. The mercury concentration in air decreased to background levels within 1.5 km of the waste ponds.
11. The total deposition of mercury near the ponds was estimated to be 7.4 to 9.0 $\mu\text{g}/\text{m}^2/\text{d}$, 15 to 20 times higher than values reported near Swedish chloralkali plants.
12. Levels of mercury in soils near the waste ponds did not appear to be abnormally elevated.
13. Mercury in grass growing directly on the waste ponds was elevated 50 times over world averages for grasses.
14. The net flux of mercury in leachates from the waste ponds to the North Fork Holston River was estimated to range from 25 to 353 g/d, with one outfall pipe from pond 1 contributing 25 to 50% of the total Hg flux to the river.

15. The estimated annual loss of mercury from the waste ponds was 36 ± 4 kg/year emitted to the atmosphere and 39 ± 2 kg/year leached into the river.
16. The total mercury burden of the waste ponds was estimated to be $\sim 10^4$ kg in the upper 2 m of waste pond 1 alone.
17. It was estimated that it would take 25 years to deplete the soluble mercury in the upper 2 m of pond 1 via leaching.
18. It was estimated that it would take 250 years to deplete the elemental Hg in the upper 2 m of pond 1 via atmospheric emission.

Distribution of Mercury in Water and Sediments
of the Holston River - Cherokee Reservoir System

1. Concentrations of mercury in water (0.15 ± 0.05 $\mu\text{g/liter}$) and on suspended matter (7.6 ± 3.8 $\mu\text{g/liter}$) immediately downstream from the plant site were typically 20 times higher than immediately upstream.
2. The silt-clay fraction of bottom sediment immediately downstream from the plant site contained up to 200 times as much mercury (19.3 ± 1.3 $\mu\text{g/g}$) as similar sediments collected immediately upstream.
3. Variations in mercury concentrations in water, suspended matter, and bottom sediment over more than 250 km of the downstream river-reservoir system indicated rapid decreases to a point ~ 40 km below the plant site followed by only slight variations.

4. Field observations and application of a simple dilution model indicated that the rapid decreases in total mercury concentrations in water as a function of distance below the plant site could be partially accounted for by dilution. However, up to 50% of the observed decreases were apparently related to factors other than dilution. Since the dilution model predicted downstream mercury concentrations most successfully at higher stream discharge, it was concluded that non-dilution processes were more important at low stream discharge.
5. Mercury in the silt-clay fraction of the surficial sediments of Cherokee Reservoir, which appears to have been an important sink for mercury discharged during plant operation, ranged from 0.47 to 2.40 $\mu\text{g/g}$. Background levels of mercury expected in sediments from this impoundment in the absence of anthropogenic inputs were estimated to be $\sim 0.2 \mu\text{g/g}$.
6. Mercury profiles in sediment cores from Cherokee Reservoir suggest that mercury input has been decreasing with time. The chronology of mercury deposition in Cherokee Reservoir inferred from ^{137}Cs activity profiles in the cores suggested that mercury input to the impoundment was highest during early operations of the upstream chloralkali plant and perhaps again near the end of operations.
7. Overall results of this part of the study substantiated that recovery of the Holston-Cherokee system formerly subjected to high discharges of mercury by the chloralkali plant has been very slow and is far from complete. Continued losses of mercury from mercury-contaminated solid wastes stored at the plant site are implicated in this slow recovery.

Distribution of Mercury in Fish and Invertebrates
of the Holston River - Cherokee Reservoir System

1. Total mercury concentration in rock bass, hog suckers, and shiners collected at select stations over 133 km of the North Fork Holston between the chloralkali waste ponds and the confluence with the Holston River near Kingsport, Tennessee, in 1974 and 1975 were above the 0.5 $\mu\text{g/g}$ FDA "action level" in effect at that time.
2. The majority of total mercury in fish muscle from the North Fork Holston River was in the methylated form.
3. Fish collected 9 and 43 km above the plant had mean mercury levels below 0.5 $\mu\text{g/g}$. Fish collected 5.5 km above the plant showed slightly elevated mercury levels, possibly reflecting migration or other diffuse sources of mercury to the river in the nearfield area above the chloralkali plant.
4. Total mercury concentration in benthic invertebrates in the North Fork below the chloralkali waste ponds downstream to Kingsport, Tennessee, followed the same general pattern as mercury levels in fish species. Total mercury concentration was highest below the ponds, then decreased downstream.
5. On the order of 50% of the mercury in benthic invertebrates was in the methylated form.
6. Total mercury concentration in axial muscle of rock bass and hog sucker was positively related to size of fish (weight) in some, but not all, of the station-date analyses performed. Inadequate sample size or the migratory nature of fish resulting in different exposure conditions may account for this discrepancy.

7. A multiple regression analysis of factors affecting total mercury concentration in fish and benthic invertebrates in the North Fork Holston River indicated 96.4% of the variance in mercury concentration in rock bass was explained by total mercury concentration in the water column, dissolved total mercury concentration in water, total mercury concentration in sediment (< 44- μ m size), and total mercury concentration in benthic invertebrates. For hog sucker, 80.5% of the variance in total mercury concentration was explained by the above four independent variables. In benthic invertebrates, 97.4% of the variance of total mercury concentration was explained by total mercury concentration in the water column, total mercury dissolved in water, and total mercury concentration in sediment (< 44- μ m size).
8. The regression analysis of mercury concentration in sediment on fish mercury concentration indicated that sediment concentrations of approximately 1 μ g/g (< 44- μ m size) may indicate fish levels could reach 0.5 μ g/g.
9. The multiple regression approach should be evaluated to determine if mercury or other element levels in fish tissue can be predicted from select variables in the environment. This approach may prove useful in large-scale monitoring programs where exhaustive data collection may not be feasible, or it could be a signal for allocation of a more intensive monitoring effort.

10. Bass and bluegill in Cherokee Reservoir generally contained total mercury in muscle below 0.5 $\mu\text{g/g}$. The majority of bass and bluegill analyzed in the reservoir that did contain mercury above 0.5 $\mu\text{g/g}$ were collected at the upper end of the reservoir. As was the case in the North Fork, some but not all analyses in Cherokee Reservoir revealed a significant relation between mercury concentration and fish weight. All shad collected in the reservoir were below 0.5 $\mu\text{g/g}$ total mercury.
11. Of striped bass collected in Cherokee Reservoir between 1972 and 1974, only fish collected in 1972 had mercury concentrations above 0.5 $\mu\text{g/g}$. Mean mercury concentration in muscle in 1973 and 1974 in striped bass was in the 0.4- to 0.5- $\mu\text{g/g}$ range.
12. The majority of mercury in Cherokee Reservoir fish muscle in the form of methylmercury.

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APPENDIX A. SUMMARY OF HOLSTON RIVER/CHEROKEE RESERVOIR FISH DATA [river km = distance above (-) and below plant, tot Hg = total mercury in axial muscle ($\mu\text{g/g}$), MeHg = methylmercury in axial muscle ($\mu\text{g/g}$), weight = fish wet weight (grams), and • represents data not measured]

YEAR	RIVER_KM	STATION	SPECIES	TOT_HG	ME_HG	WEIGHT
1974	-5.5	MCCRADY	ROCK BASS	0.460	.	79.1
1974	-5.5	MCCRADY	ROCK BASS	1.360	0.840	111.7
1974	-5.5	MCCRADY	ROCK BASS	0.370	0.270	41.5
1974	-5.5	MCCRADY	ROCK BASS	0.195	0.165	18.0
1974	-5.5	MCCRADY	ROCK BASS	0.430	0.400	164.7
1974	-5.5	MCCRADY	ROCK BASS	0.820	.	216.5
1974	-5.5	MCCRADY	ROCK BASS	1.320	1.240	219.0
1974	-5.5	MCCRADY	ROCK BASS	0.980	.	185.8
1974	-5.5	MCCRADY	ROCK BASS	1.080	0.870	303.2
1974	-5.5	MCCRADY	ROCK BASS	0.220	0.200	7.8
1974	-5.5	MCCRADY	ROCK BASS	0.930	0.990	96.6
1974	-5.5	MCCRADY	ROCK BASS	0.300	0.300	15.1
1974	-5.5	MCCRADY	ROCK BASS	0.690	0.570	72.7
1974	-5.5	MCCRADY	HOG SUCKER	0.450	0.480	157.9
1974	-5.5	MCCRADY	HOG SUCKER	0.600	0.660	191.0
1974	-5.5	MCCRADY	HOG SUCKER	0.750	0.820	103.4
1974	-5.5	MCCRADY	HOG SUCKER	0.370	0.390	129.0
1974	-5.5	MCCRADY	HOG SUCKER	1.130	1.170	70.0
1974	-5.5	MCCRADY	HOG SUCKER	0.750	.	150.9
1974	-5.5	MCCRADY	HOG SUCKER	0.210	0.220	27.7
1974	-5.5	MCCRADY	HOG SUCKER	0.340	.	142.3
1974	-5.5	MCCRADY	HOG SUCKER	0.150	0.150	12.1
1974	-5.5	MCCRADY	HOG SUCKER	0.180	0.170	11.8
1974	-5.5	MCCRADY	HOG SUCKER	0.180	0.180	14.3
1974	-5.5	MCCRADY	COMMON SHINER	0.270	0.150	28.9
1974	-5.5	MCCRADY	COMMON SHINER	0.220	.	13.7
1974	-5.5	MCCRADY	COMMON SHINER	0.165	0.165	14.1
1974	-5.5	MCCRADY	COMMON SHINER	0.220	0.175	16.0
1974	-5.5	MCCRADY	COMMON SHINER	0.230	0.150	20.5
1974	-5.5	MCCRADY	COMMON SHINER	0.500	0.320	47.1
1974	-5.5	MCCRADY	COMMON SHINER	0.150	.	11.8
1974	-5.5	MCCRADY	COMMON SHINER	0.260	.	10.9
1974	-5.5	MCCRADY	COMMON SHINER	0.200	0.200	27.3
1974	-5.5	MCCRADY	COMMON SHINER	0.200	.	7.0
1974	-5.5	MCCRADY	COMMON SHINER	0.290	0.250	12.5
1974	-5.5	MCCRADY	COMMON SHINER	0.185	.	10.8
1974	-5.5	MCCRADY	COMMON SHINER	0.270	.	11.5
1974	-5.5	MCCRADY	COMMON SHINER	0.234	.	10.3
1974	-5.5	MCCRADY	COMMON SHINER	0.380	.	11.6
1974	-5.5	MCCRADY	COMMON SHINER	0.260	0.260	10.1
1974	-5.5	MCCRADY	COMMON SHINER	0.300	.	7.7
1974	-5.5	MCCRADY	COMMON SHINER	0.300	0.270	8.0
1974	-5.5	MCCRADY	COMMON SHINER	0.240	.	8.4
1974	-5.5	MCCRADY	COMMON SHINER	0.670	0.700	7.0
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	1.140	0.930	28.3
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	0.890	0.860	19.6
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	1.450	1.280	48.1
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	1.380	.	27.3
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	1.660	.	26.2
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	2.300	.	28.2
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	1.550	.	20.2
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	1.39	1.18	23.5
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	1.18	0.98	27.7
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	1.79	1.18	78.2
1974	3.7	BELOW DISPOSAL PONDS	ROCK BASS	2.80	2.40	184.3
1974	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	1.53	1.46	113.2
1974	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	1.46	1.13	41.2
1974	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	1.27	1.27	111.9

APPENDIX A. (continued)

YEAR	RIVER_KM	STATION	SPECIES	TOT_HG	ME_HG	WEIGHT
1974	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	1.80	1.98	1.6
1974	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	0.93	0.97	3.7
1974	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	1.77	1.79	1.8
1974	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	2.70	.	1.6
1974	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	1.72	.	3.3
1974	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	0.50	.	2.4
1974	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	2.45	.	2.4
1974	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	1.71	.	2.1
1974	21.0	HAYTERS GAP	ROCK BASS	1.15	1.18	11.8
1974	21.0	HAYTERS GAP	ROCK BASS	1.14	1.09	8.2
1974	21.0	HAYTERS GAP	ROCK BASS	1.43	1.47	3.8
1974	21.0	HAYTERS GAP	ROCK BASS	2.30	2.20	96.4
1974	21.0	HAYTERS GAP	ROCK BASS	1.04	.	11.0
1974	21.0	HAYTERS GAP	ROCK BASS	0.94	.	10.0
1974	21.0	HAYTERS GAP	ROCK BASS	1.26	.	13.7
1974	21.0	HAYTERS GAP	ROCK BASS	0.90	.	12.6
1974	21.0	HAYTERS GAP	ROCK BASS	1.14	.	5.7
1974	21.0	HAYTERS GAP	HUCK BASS	0.44	0.47	6.9
1974	21.0	HAYTERS GAP	ROCK BASS	0.91	0.85	13.2
1974	21.0	HAYTERS GAP	ROCK BASS	1.10	.	10.2
1974	21.0	HAYTERS GAP	ROCK BASS	1.09	.	11.7
1974	21.0	HAYTERS GAP	ROCK BASS	0.79	.	7.7
1974	21.0	HAYTERS GAP	ROCK BASS	1.32	.	8.4
1974	21.0	HAYTERS GAP	ROCK BASS	1.25	.	6.2
1974	21.0	HAYTERS GAP	HOG SUCKER	1.16	1.08	87.5
1974	21.0	HAYTERS GAP	HOG SUCKER	1.39	1.08	48.1
1974	21.0	HAYTERS GAP	HOG SUCKER	1.12	1.01	52.6
1974	21.0	HAYTERS GAP	HOG SUCKER	1.03	0.89	45.0
1974	21.0	HAYTERS GAP	HOG SUCKER	1.28	1.19	164.1
1974	21.0	HAYTERS GAP	HOG SUCKER	1.43	1.11	112.0
1974	21.0	HAYTERS GAP	COMMON SHINER	0.70	0.50	11.8
1974	21.0	HAYTERS GAP	COMMON SHINER	1.08	0.96	3.6
1974	21.0	HAYTERS GAP	COMMON SHINER	0.95	0.67	4.0
1974	21.0	HAYTERS GAP	COMMON SHINER	0.64	0.61	22.2
1974	21.0	HAYTERS GAP	COMMON SHINER	0.85	0.79	11.6
1974	21.0	HAYTERS GAP	COMMON SHINER	1.28	.	2.6
1974	21.0	HAYTERS GAP	COMMON SHINER	0.68	.	4.2
1974	21.0	HAYTERS GAP	COMMON SHINER	1.07	.	2.5
1974	21.0	HAYTERS GAP	COMMON SHINER	0.73	.	1.7
1974	21.0	HAYTERS GAP	COMMON SHINER	1.21	.	2.2
1974	21.0	HAYTERS GAP	COMMON SHINER	1.70	.	2.2
1974	21.0	HAYTERS GAP	COMMON SHINER	1.36	.	2.5
1974	21.0	HAYTERS GAP	COMMON SHINER	1.21	.	1.5
1974	21.0	HAYTERS GAP	COMMON SHINER	1.30	.	1.3
1974	133	KINGSPORT	ROCK BASS	2.500	1.630	156.9
1974	133	KINGSPORT	ROCK BASS	1.820	.	150.6
1974	133	KINGSPORT	ROCK BASS	0.540	0.510	23.6
1974	133	KINGSPORT	ROCK BASS	0.560	.	47.3
1974	133	KINGSPORT	ROCK BASS	1.610	.	118.0
1974	133	KINGSPORT	ROCK BASS	0.108	.	43.0
1974	133	KINGSPORT	ROCK BASS	0.610	.	25.0
1974	133	KINGSPORT	ROCK BASS	0.930	.	121.2
1974	133	KINGSPORT	ROCK BASS	2.020	1.570	141.0
1974	133	KINGSPORT	ROCK BASS	1.920	.	139.0
1974	133	KINGSPORT	HOG SUCKER	0.485	0.480	55.6
1974	133	KINGSPORT	HOG SUCKER	0.370	0.260	53.4
1974	133	KINGSPORT	COMMON SHINER	0.810	.	7.9
1974	133	KINGSPORT	COMMON SHINER	0.470	0.380	1.7
1974	133	KINGSPORT	COMMON SHINER	1.170	0.800	2.7
1974	133	KINGSPORT	COMMON SHINER	1.130	0.910	3.2
1974	133	KINGSPORT	COMMON SHINER	0.875	0.800	2.5
1974	133	KINGSPORT	COMMON SHINER	0.560	0.450	1.6
1974	133	KINGSPORT	COMMON SHINER	0.810	0.550	4.0
1974	133	KINGSPORT	COMMON SHINER	0.460	.	2.8
1974	133	KINGSPORT	COMMON SHINER	0.740	.	6.4
1974	133	KINGSPORT	COMMON SHINER	0.970	.	2.0
1974	133	KINGSPORT	COMMON SHINER	0.600	.	1.3
1974	133	KINGSPORT	COMMON SHINER	0.350	.	3.4
1974	133	KINGSPORT	COMMON SHINER	1.080	.	1.2

APPENDIX A. (continued)

YEAR	RIVER_KM	STATION	SPECIES	TOT_HG	ME_HG	WEIGHT
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.460	-	75.8
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.470	0.420	41.0
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.460	0.430	54.7
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.220	.	58.0
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.192	.	46.0
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.180	0.175	79.6
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.390	0.320	35.0
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.430	0.440	63.0
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.300	.	45.1
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.330	0.190	73.5
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.130	0.115	123.3
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.240	.	64.5
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.380	0.097	90.7
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.136	.	107.4
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.130	.	74.0
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.540	.	117.0
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.520	0.510	156.3
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.340	.	65.3
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.105	0.094	112.1
1974	196	H.B.DAY BRIDGE	BLUEGILL	0.540	0.480	105.8
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.093	.	65.0
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.082	.	70.5
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.066	.	45.6
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.038	.	2.1
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.080	.	43.6
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.097	.	60.8
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.051	.	51.2
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.110	.	56.4
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.053	.	52.7
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.046	0.016	4.1
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.140	0.135	6.2
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.054	0.037	44.5
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.170	0.120	25.8
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.071	0.053	39.6
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.080	0.070	63.7
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.080	0.074	71.3
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.083	0.066	77.9
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.125	0.120	227.0
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.105	0.066	232.0
1974	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.150	0.130	284.0
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.180	0.140	176.6
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.230	0.200	284.5
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.260	.	236.7
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.290	0.280	324.6
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.160	0.165	129.6
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	1.160	0.890	2072.7
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.190	0.185	133.8
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.420	0.390	908.4
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.250	0.250	229.0
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.280	.	229.8
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.520	0.460	891.4
1974	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.200	0.210	213.8
1974	196	H.B.DAY BRIDGE	SMALLMOUTH BASS	1.640	1.170	1477.3
1974	196	H.B.DAY BRIDGE	SMALLMOUTH BASS	0.240	0.200	272.5
1974	196	H.B.DAY BRIDGE	SMALLMOUTH BASS	0.220	0.120	441.5
1974	196	H.B.DAY BRIDGE	SMALLMOUTH BASS	0.210	0.170	233.1
1974	196	H.B.DAY BRIDGE	SMALLMOUTH BASS	0.260	0.220	209.5
1974	196	H.B.DAY BRIDGE	SMALLMOUTH BASS	0.430	0.430	660.3
1974	196	H.B.DAY BRIDGE	SMALLMOUTH BASS	0.530	0.450	35.3
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.039	0.023	2.1
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.064	0.050	2.1
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.048	0.025	2.2
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.043	0.025	2.6
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.066	0.047	1.5
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.031	0.014	1.3

APPENDIX A. (continued)

YEAR	RIVER_KM	STATION	SPECIES	TOT_HG	ME_HG	WEIGHT
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.034	0.015	2.0
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.050	0.025	1.7
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.064	0.039	4.4
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.040	0.024	1.6
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.037	.	1.2
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.068	.	1.2
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.040	.	2.0
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.034	.	2.0
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.046	.	1.3
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.067	.	1.3
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.040	.	1.2
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.044	.	1.2
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.054	.	1.2
1974	196	H.B.DAY BRIDGE	THREADFIN SHAD	0.044	.	1.1
1974	238	25E BRIDGE	BLUEGILL	0.200	.	65.9
1974	238	25E BRIDGE	BLUEGILL	0.086	.	31.0
1974	238	25E BRIDGE	BLUEGILL	0.341	.	73.9
1974	238	25E BRIDGE	BLUEGILL	0.088	.	28.1
1974	238	25E BRIDGE	BLUEGILL	0.066	.	47.0
1974	238	25E BRIDGE	BLUEGILL	0.210	0.200	74.5
1974	238	25E BRIDGE	BLUEGILL	0.084	0.070	30.5
1974	238	25E BRIDGE	BLUEGILL	0.230	.	45.0
1974	238	25E BRIDGE	BLUEGILL	0.190	0.170	57.7
1974	238	25E BRIDGE	BLUEGILL	0.056	0.036	11.4
1974	238	25E BRIDGE	BLUEGILL	0.129	.	33.1
1974	238	25E BRIDGE	BLUEGILL	0.077	.	31.3
1974	238	25E BRIDGE	BLUEGILL	0.110	0.064	16.1
1974	238	25E BRIDGE	BLUEGILL	0.090	0.047	60.6
1974	238	25E BRIDGE	BLUEGILL	0.126	.	48.3
1974	238	25E BRIDGE	BLUEGILL	0.092	0.043	35.4
1974	238	25E BRIDGE	BLUEGILL	0.070	0.068	25.3
1974	238	25E BRIDGE	BLUEGILL	0.089	0.057	43.5
1974	238	25E BRIDGE	BLUEGILL	0.067	0.057	23.3
1974	238	25E BRIDGE	BLUEGILL	0.142	.	33.4
1974	238	25E BRIDGE	GIZZARD SHAD	0.066	0.047	39.1
1974	238	25E BRIDGE	GIZZARD SHAD	0.069	0.027	44.0
1974	238	25E BRIDGE	GIZZARD SHAD	0.053	0.032	32.5
1974	238	25E BRIDGE	GIZZARD SHAD	0.080	.	38.5
1974	238	25E BRIDGE	GIZZARD SHAD	0.046	0.037	133.8
1974	238	25E BRIDGE	GIZZARD SHAD	0.061	.	85.5
1974	238	25E BRIDGE	GIZZARD SHAD	0.065	.	44.0
1974	238	25E BRIDGE	GIZZARD SHAD	0.061	0.052	63.2
1974	238	25E BRIDGE	GIZZARD SHAD	0.039	0.028	115.6
1974	238	25E BRIDGE	GIZZARD SHAD	0.052	0.038	85.2
1974	238	25E BRIDGE	GIZZARD SHAD	0.053	.	45.9
1974	238	25E BRIDGE	GIZZARD SHAD	0.075	.	41.8
1974	238	25E BRIDGE	GIZZARD SHAD	0.058	.	43.9
1974	238	25E BRIDGE	GIZZARD SHAD	0.111	.	41.7
1974	238	25E BRIDGE	GIZZARD SHAD	0.063	.	36.6
1974	238	25E BRIDGE	GIZZARD SHAD	0.026	0.004	3.0
1974	238	25E BRIDGE	GIZZARD SHAD	0.023	0.009	2.4
1974	238	25E BRIDGE	GIZZARD SHAD	0.025	.	3.3
1974	238	25E BRIDGE	GIZZARD SHAD	0.076	.	47.0
1974	238	25E BRIDGE	GIZZARD SHAD	0.070	0.057	77.7
1974	238	25E BRIDGE	LARGemouth BASS	0.105	.	269.5
1974	238	25E BRIDGE	LARGemouth BASS	0.085	0.055	26.1
1974	238	25E BRIDGE	LARGemouth BASS	0.105	0.078	19.3
1974	238	25E BRIDGE	LARGemouth BASS	0.084	0.075	178.3
1974	238	25E BRIDGE	LARGemouth BASS	0.042	.	21.7
1974	238	25E BRIDGE	LARGemouth BASS	0.120	0.071	31.0
1974	238	25E BRIDGE	LARGemouth BASS	0.061	.	28.0
1974	238	25E BRIDGE	LARGemouth BASS	0.062	.	21.9
1974	238	25E BRIDGE	LARGemouth BASS	0.059	.	32.8

APPENDIX A. (continued)

YEAR	RIVER_KM	STATION	SPECIES	TOT_HG	ME_HG	WEIGHT
1974	238	25E BRIDGE	LARGEMOUTH BASS	0.110	0.091	281.6
1974	238	25E BRIDGE	LARGEMOUTH BASS	0.290	0.250	726.0
1974	238	25E BRIDGE	LARGEMOUTH BASS	0.109	.	230.0
1974	238	25E BRIDGE	LARGEMOUTH BASS	0.097	0.084	363.7
1974	238	25E BRIDGE	LARGEMOUTH BASS	0.058	0.052	117.6
1974	238	25E BRIDGE	LARGEMOUTH BASS	0.096	0.080	211.5
1974	238	25E BRIDGE	LARGEMOUTH BASS	0.060	.	28.6
1974	238	25E BRIDGE	LARGEMOUTH BASS	0.084	.	39.1
1974	238	25E BRIDGE	LARGEMOUTH BASS	0.055	0.032	46.5
1974	238	25E BRIDGE	LARGEMOUTH BASS	0.078	.	32.2
1974	238	25E BRIDGE	SMALLMOUTH BASS	0.125	0.120	208.1
1974	238	25E BRIDGE	SMALLMOUTH BASS	0.055	0.040	6.5
1974	238	25E BRIDGE	SMALLMOUTH BASS	0.155	0.110	103.2
1974	238	25E BRIDGE	THREADFIN SHAD	0.029	0.009	2.0
1974	238	25E BRIDGE	THREADFIN SHAD	0.028	0.023	0.9
1974	238	25E BRIDGE	THREADFIN SHAD	0.024	0.010	2.0
1974	238	25E BRIDGE	THREADFIN SHAD	0.028	0.014	1.0
1974	238	25E BRIDGE	THREADFIN SHAD	0.026	0.011	2.0
1974	238	25E BRIDGE	THREADFIN SHAD	0.019	0.009	1.6
1974	238	25E BRIDGE	THREADFIN SHAD	0.025	0.018	0.8
1974	238	25E BRIDGE	THREADFIN SHAD	0.022	0.008	1.2
1974	238	25E BRIDGE	THREADFIN SHAD	0.014	0.012	1.4
1974	238	25E BRIDGE	THREADFIN SHAD	0.023	.	1.0
1974	238	25E BRIDGE	THREADFIN SHAD	0.027	.	0.7
1974	238	25E BRIDGE	THREADFIN SHAD	0.039	.	1.9
1974	238	25E BRIDGE	THREADFIN SHAD	0.018	.	1.3
1974	238	25E BRIDGE	THREADFIN SHAD	0.027	.	2.3
1974	238	25E BRIDGE	THREADFIN SHAD	0.024	.	1.0
1974	238	25E BRIDGE	THREADFIN SHAD	0.023	.	1.0
1974	238	25E BRIDGE	THREADFIN SHAD	0.028	.	2.4
1974	238	25E BRIDGE	THREADFIN SHAD	0.023	.	1.0
1974	238	25E BRIDGE	THREADFIN SHAD	0.100	.	15.0
1974	238	25E BRIDGE	THREADFIN SHAD	0.034	.	0.5
1974	277	CHEROKEE DAM	BLUEGILL	0.270	.	46.9
1974	277	CHEROKEE DAM	BLUEGILL	0.128	0.099	6.9
1974	277	CHEROKEE DAM	BLUEGILL	0.060	.	12.5
1974	277	CHEROKEE DAM	BLUEGILL	0.098	.	9.2
1974	277	CHEROKEE DAM	BLUEGILL	0.290	0.220	10.0
1974	277	CHEROKEE DAM	BLUEGILL	0.116	.	9.9
1974	277	CHEROKEE DAM	BLUEGILL	0.161	0.068	12.6
1974	277	CHEROKEE DAM	BLUEGILL	0.111	.	62.7
1974	277	CHEROKEE DAM	BLUEGILL	0.194	0.140	39.5
1974	277	CHEROKEE DAM	BLUEGILL	0.084	.	17.9
1974	277	CHEROKEE DAM	BLUEGILL	0.109	.	17.9
1974	277	CHEROKEE DAM	BLUEGILL	0.255	0.170	74.3
1974	277	CHEROKEE DAM	BLUEGILL	0.158	0.081	103.3
1974	277	CHEROKEE DAM	BLUEGILL	0.124	0.089	69.9
1974	277	CHEROKEE DAM	BLUEGILL	0.380	0.300	48.8
1974	277	CHEROKEE DAM	BLUEGILL	0.114	.	17.5
1974	277	CHEROKEE DAM	BLUEGILL	0.090	.	11.5
1974	277	CHEROKEE DAM	BLUEGILL	0.079	.	17.0
1974	277	CHEROKEE DAM	BLUEGILL	0.125	0.068	18.3
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.051	0.031	50.1
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.078	.	59.9
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.048	.	45.5
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.068	0.050	20.3
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.041	.	49.2
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.075	0.055	170.7
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.058	.	49.4
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.035	0.034	18.0
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.049	.	55.9
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.059	.	67.0
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.036	0.020	55.2
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.056	.	55.5
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.059	0.040	71.7

APPENDIX A. (continued)

YEAR	RIVER_KM	STATION	SPECIES	TOT_HG	ME_HG	WEIGHT
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.046	.	42.6
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.061	0.037	60.1
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.054	.	46.6
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.064	0.045	174.5
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.058	0.033	129.2
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.048	.	113.1
1974	277	CHEROKEE DAM	GIZZARD SHAD	0.047	0.040	43.9
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.400	0.220	450.6
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.184	0.170	226.5
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.210	0.120	181.0
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.166	.	107.9
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.161	.	179.8
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.194	0.130	261.4
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.095	0.059	144.0
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.310	0.160	352.7
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.120	0.092	103.2
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.068	0.046	89.5
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.169	0.150	57.7
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.210	0.140	200.0
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.230	.	257.3
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.280	.	247.6
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.109	.	87.3
1974	277	CHEROKEE DAM	LARGEMOUTH BASS	0.130	.	144.7
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.082	0.055	17.5
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.061	0.053	24.7
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.099	0.090	20.0
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.084	0.074	10.8
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.048	0.040	12.1
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.051	0.042	10.3
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.116	0.086	48.7
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.016	.	1.6
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.052	.	1.3
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.028	.	1.9
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.027	.	2.3
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.025	.	2.2
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.031	.	1.6
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.029	.	2.4
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.031	.	2.4
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.034	.	1.9
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.024	.	1.9
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.038	.	1.9
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.033	.	1.5
1974	277	CHEROKEE DAM	THREADFIN SHAD	0.026	.	1.2
1975	-43	CHATHAM HILL	ROCK BASS	0.540	0.500	101.2
1975	-43	CHATHAM HILL	ROCK BASS	0.250	0.250	82.2
1975	-43	CHATHAM HILL	ROCK BASS	0.280	0.240	73.8
1975	-43	CHATHAM HILL	ROCK BASS	0.240	0.240	64.9
1975	-43	CHATHAM HILL	ROCK BASS	0.180	0.160	40.7
1975	-43	CHATHAM HILL	ROCK BASS	0.180	0.170	47.0
1975	-43	CHATHAM HILL	ROCK BASS	0.180	0.170	31.4
1975	-43	CHATHAM HILL	ROCK BASS	0.092	0.082	21.2
1975	-43	CHATHAM HILL	ROCK BASS	0.130	0.130	14.6
1975	-43	CHATHAM HILL	ROCK BASS	0.120	0.120	11.8
1975	-43	CHATHAM HILL	HOG SUCKER	0.310	0.270	198.6
1975	-43	CHATHAM HILL	HOG SUCKER	0.250	0.240	77.4
1975	-43	CHATHAM HILL	HOG SUCKER	0.150	0.140	25.1
1975	-43	CHATHAM HILL	HOG SUCKER	0.110	0.110	32.4
1975	-43	CHATHAM HILL	HOG SUCKER	0.100	0.100	19.3
1975	-43	CHATHAM HILL	HOG SUCKER	0.130	0.130	15.2
1975	-43	CHATHAM HILL	HOG SUCKER	0.160	0.160	62.5
1975	-43	CHATHAM HILL	HOG SUCKER	0.200	0.190	98.8
1975	-43	CHATHAM HILL	HOG SUCKER	0.160	0.150	132.8
1975	-43	CHATHAM HILL	HOG SUCKER	0.370	0.350	282.9
1975	-43	CHATHAM HILL	COMMON SHINER	0.170	0.160	7.0
1975	-43	CHATHAM HILL	COMMON SHINER	0.180	0.160	7.7

APPENDIX A. (continued)

YEAR	RIVER_KM	STATION	SPECIES	TOT_HG	ME_HG	WEIGHT
1975	-43	CHATHAM HILL	COMMON SHINER	0.120	0.110	8.9
1975	-43	CHATHAM HILL	COMMON SHINER	0.088	0.083	8.2
1975	-43	CHATHAM HILL	COMMON SHINER	0.088	0.084	9.9
1975	-9	NEIL BRIDGE	ROCK BASS	0.190	.	9.3
1975	-9	NEIL BRIDGE	ROCK BASS	0.200	.	17.2
1975	-9	NEIL BRIDGE	ROCK BASS	0.220	.	17.4
1975	-9	NEIL BRIDGE	ROCK BASS	0.320	.	28.2
1975	-9	NEIL BRIDGE	ROCK BASS	0.210	.	50.3
1975	-9	NEIL BRIDGE	ROCK BASS	0.670	.	56.3
1975	-9	NEIL BRIDGE	ROCK BASS	0.380	.	69.6
1975	-9	NEIL BRIDGE	ROCK BASS	0.340	.	108.6
1975	-9	NEIL BRIDGE	ROCK BASS	0.380	.	104.2
1975	-9	NEIL BRIDGE	ROCK BASS	0.310	.	133.9
1975	-9	NEIL BRIDGE	HOG SUCKER	0.110	.	15.3
1975	-9	NEIL BRIDGE	HOG SUCKER	0.170	.	28.8
1975	-9	NEIL BRIDGE	HOG SUCKER	0.150	.	48.5
1975	-9	NEIL BRIDGE	HOG SUCKER	0.150	.	51.8
1975	-9	NEIL BRIDGE	HOG SUCKER	0.190	.	96.5
1975	-9	NEIL BRIDGE	HOG SUCKER	0.200	.	117.9
1975	-9	NEIL BRIDGE	HOG SUCKER	0.220	.	143.3
1975	-9	NEIL BRIDGE	HOG SUCKER	0.670	.	189.5
1975	-9	NEIL BRIDGE	HOG SUCKER	0.240	.	210.9
1975	-9	NEIL BRIDGE	HOG SUCKER	0.680	.	274.2
1975	-9.0	NEIL BRIDGE	COMMON SHINER	0.120	.	7.8
1975	-9.0	NEIL BRIDGE	COMMON SHINER	0.120	.	9.9
1975	-9.0	NEIL BRIDGE	COMMON SHINER	0.110	.	10.1
1975	-9.0	NEIL BRIDGE	COMMON SHINER	0.170	.	10.1
1975	-9.0	NEIL BRIDGE	COMMON SHINER	0.085	.	12.0
1975	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	1.770	.	51.0
1975	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	1.340	.	36.3
1975	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	1.810	.	63.5
1975	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	2.500	.	87.6
1975	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	2.500	.	101.1
1975	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	2.700	.	143.6
1975	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	1.840	.	227.0
1975	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	1.690	.	226.5
1975	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	0.950	.	311.4
1975	3.7	BELOW DISPOSAL PONDS	HOG SUCKER	4.300	.	267.9
1975	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	2.200	.	2.1
1975	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	2.500	.	2.0
1975	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	0.710	.	2.7
1975	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	2.700	.	1.8
1975	3.7	BELOW DISPOSAL PONDS	COMMON SHINER	2.500	.	1.7
1975	9.7	MCKENNA ISLAND	ROCK BASS	2.100	2.00	23.8
1975	9.7	MCKENNA ISLAND	ROCK BASS	2.100	1.86	40.3
1975	9.7	MCKENNA ISLAND	ROCK BASS	2.100	1.57	33.6
1975	9.7	MCKENNA ISLAND	ROCK BASS	1.480	1.38	40.8
1975	9.7	MCKENNA ISLAND	ROCK BASS	1.340	1.01	38.3
1975	9.7	MCKENNA ISLAND	ROCK BASS	1.400	1.15	52.3
1975	9.7	MCKENNA ISLAND	ROCK BASS	1.280	0.98	70.8
1975	9.7	MCKENNA ISLAND	ROCK BASS	2.600	2.40	72.0
1975	9.7	MCKENNA ISLAND	ROCK BASS	1.340	1.30	86.1
1975	9.7	MCKENNA ISLAND	ROCK BASS	2.400	2.30	91.6
1975	9.7	MCKENNA ISLAND	HOG SUCKER	1.700	1.70	26.5
1975	9.7	MCKENNA ISLAND	HOG SUCKER	1.720	1.47	28.1
1975	9.7	MCKENNA ISLAND	HOG SUCKER	1.530	1.27	26.7
1975	9.7	MCKENNA ISLAND	HOG SUCKER	1.900	1.85	84.1
1975	9.7	MCKENNA ISLAND	HOG SUCKER	2.600	2.50	119.7
1975	9.7	MCKENNA ISLAND	HOG SUCKER	2.500	2.40	147.6
1975	9.7	MCKENNA ISLAND	HOG SUCKER	2.750	2.79	164.8
1975	9.7	MCKENNA ISLAND	HOG SUCKER	2.000	1.93	259.8
1975	9.7	MCKENNA ISLAND	HOG SUCKER	2.500	2.10	256.8
1975	9.7	MCKENNA ISLAND	HOG SUCKER	2.700	2.70	536.1

APPENDIX A. (continued)

YEAR	RIVER_KM	STATION	SPECIES	TOT_HG	ME_HG	WEIGHT
1975	9.7	MCKENNA ISLAND	COMMON SHINER	2.400	1.98	2.4
1975	9.7	MCKENNA ISLAND	COMMON SHINER	2.300	1.97	2.5
1975	9.7	MCKENNA ISLAND	COMMON SHINER	2.400	1.98	2.7
1975	9.7	MCKENNA ISLAND	COMMON SHINER	2.100	1.84	2.8
1975	9.7	MCKENNA ISLAND	COMMON SHINER	2.500	.	2.7
1975	21.0	HAYTERS GAP	HOG SUCKER	1.290	.	37.4
1975	21.0	HAYTERS GAP	HOG SUCKER	1.460	.	42.6
1975	21.0	HAYTERS GAP	HOG SUCKER	1.430	.	74.3
1975	21.0	HAYTERS GAP	HOG SUCKER	1.720	.	73.7
1975	21.0	HAYTERS GAP	HOG SUCKER	1.460	.	103.7
1975	21.0	HAYTERS GAP	HOG SUCKER	1.600	.	147.6
1975	21	HAYTERS GAP	HOG SUCKER	2.200	.	148.1
1975	21	HAYTERS GAP	HOG SUCKER	2.300	.	223.7
1975	21	HAYTERS GAP	HOG SUCKER	1.650	.	193.6
1975	21	HAYTERS GAP	HOG SUCKER	1.500	.	250.3
1975	21	HAYTERS GAP	COMMON SHINER	2.300	.	9.2
1975	21	HAYTERS GAP	COMMON SHINER	0.990	.	11.2
1975	21	HAYTERS GAP	COMMON SHINER	1.220	.	10.4
1975	21	HAYTERS GAP	COMMON SHINER	1.360	.	9.9
1975	21	HAYTERS GAP	COMMON SHINER	0.850	.	9.3
1975	43	HINES ISLAND	ROCK BASS	1.210	.	6.1
1975	43	HINES ISLAND	ROCK BASS	1.010	.	9.1
1975	43	HINES ISLAND	ROCK BASS	1.350	.	10.6
1975	43	HINES ISLAND	ROCK BASS	0.960	.	19.6
1975	43	HINES ISLAND	ROCK BASS	1.400	.	27.8
1975	43	HINES ISLAND	ROCK BASS	1.230	.	43.8
1975	43	HINES ISLAND	ROCK BASS	1.210	.	51.3
1975	43	HINES ISLAND	ROCK BASS	1.710	.	83.4
1975	43	HINES ISLAND	ROCK BASS	0.990	.	88.1
1975	43	HINES ISLAND	ROCK BASS	1.850	.	165.4
1975	43	HINES ISLAND	HOG SUCKER	0.730	.	18.2
1975	43	HINES ISLAND	HOG SUCKER	0.950	.	32.7
1975	43	HINES ISLAND	HOG SUCKER	1.090	.	70.1
1975	43	HINES ISLAND	HOG SUCKER	0.920	.	73.7
1975	43	HINES ISLAND	HOG SUCKER	1.220	.	134.9
1975	43	HINES ISLAND	HOG SUCKER	0.910	.	89.1
1975	43	HINES ISLAND	HOG SUCKER	1.400	.	205.6
1975	43	HINES ISLAND	HOG SUCKER	1.640	.	250.4
1975	43	HINES ISLAND	HOG SUCKER	1.190	.	455.9
1975	43	HINES ISLAND	HOG SUCKER	1.050	.	471.1
1975	43	HINES ISLAND	COMMON SHINER	1.330	.	13.2
1975	43	HINES ISLAND	COMMON SHINER	2.200	.	8.3
1975	43	HINES ISLAND	COMMON SHINER	1.160	.	9.9
1975	43	HINES ISLAND	COMMON SHINER	.	.	12.0
1975	43	HINES ISLAND	COMMON SHINER	1.030	.	10.1
1975	133	KINGSPORT	ROCK BASS	0.065	.	42.0
1975	133	KINGSPORT	ROCK BASS	0.830	.	42.6
1975	133	KINGSPORT	ROCK BASS	0.720	.	42.0
1975	133	KINGSPORT	ROCK BASS	1.420	.	42.6
1975	133	KINGSPORT	ROCK BASS	0.580	.	45.0
1975	133	KINGSPORT	ROCK BASS	0.590	.	81.4
1975	133	KINGSPORT	ROCK BASS	2.100	.	59.1
1975	133	KINGSPORT	ROCK BASS	0.120	.	60.8
1975	133	KINGSPORT	ROCK BASS	2.300	.	112.7
1975	133	KINGSPORT	ROCK BASS	2.500	.	101.2
1975	133	KINGSPORT	HOG SUCKER	0.820	.	385.4
1975	133	KINGSPORT	HOG SUCKER	1.750	.	314.4
1975	133	KINGSPORT	HOG SUCKER	2.000	.	379.5
1975	133	KINGSPORT	HOG SUCKER	0.510	.	216.8
1975	133	KINGSPORT	HOG SUCKER	0.530	.	224.2
1975	133	KINGSPORT	HOG SUCKER	0.430	.	227.2
1975	133	KINGSPORT	HOG SUCKER	0.540	.	111.8

APPENDIX A. (continued)

YEAR	RIVER_KM	STATION	SPECIES	TOT_HG	ME_HG	WEIGHT
1975	133	KINGSPORT	HOG SUCKER	0.610	.	126.3
1975	133	KINGSPORT	HOG SUCKER	0.540	.	127.8
1975	133	KINGSPORT	HOG SUCKER	0.460	.	48.6
1975	133	KINGSPORT	COMMON SHINER	0.120	.	8.4
1975	133	KINGSPORT	COMMON SHINER	0.130	.	12.0
1975	133	KINGSPORT	COMMON SHINER	1.130	.	12.3
1975	133	KINGSPORT	COMMON SHINER	1.160	.	8.4
1975	133	KINGSPORT	COMMON SHINER	0.038	.	11.3
1975	196	H.B.DAY BRIDGE	BLUEGILL	.	.	9.9
1975	196	H.B.DAY BRIDGE	BLUEGILL	0.140	0.140	5.7
1975	196	H.B.DAY BRIDGE	BLUEGILL	0.150	0.140	34.4
1975	196	H.B.DAY BRIDGE	BLUEGILL	0.130	0.120	31.4
1975	196	H.B.DAY BRIDGE	BLUEGILL	0.091	0.085	30.5
1975	196	H.B.DAY BRIDGE	BLUEGILL	0.460	0.400	51.0
1975	196	H.B.DAY BRIDGE	BLUEGILL	0.130	0.120	50.8
1975	196	H.B.DAY BRIDGE	BLUEGILL	0.130	0.120	89.3
1975	196	H.B.DAY BRIDGE	BLUEGILL	0.140	0.130	108.9
1975	196	H.B.DAY BRIDGE	BLUEGILL	0.540	0.520	102.1
1975	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.020	0.020	53.1
1975	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.130	0.100	39.3
1975	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.023	0.012	39.7
1975	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.072	0.060	24.6
1975	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.074	0.045	54.3
1975	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.023	0.015	35.5
1975	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.082	0.066	43.2
1975	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.140	0.096	38.6
1975	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.031	0.017	39.8
1975	196	H.B.DAY BRIDGE	GIZZARD SHAD	0.021	0.014	37.5
1975	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.970	0.086	2721.5
1975	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.530	0.440	1632.9
1975	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.200	0.180	547.0
1975	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.264	0.270	325.1
1975	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.220	0.220	139.1
1975	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.130	0.130	30.5
1975	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.150	0.140	24.2
1975	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.120	0.110	15.0
1975	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.110	0.100	12.1
1975	196	H.B.DAY BRIDGE	LARGEMOUTH BASS	0.220	0.200	5.2
1975	277	CHEROKEE DAM	BLUEGILL	0.063	.	15.9
1975	277	CHEROKEE DAM	BLUEGILL	0.089	.	11.6
1975	277	CHEROKEE DAM	BLUEGILL	0.062	.	16.7
1975	277	CHEROKEE DAM	BLUEGILL	0.100	.	42.8
1975	277	CHEROKEE DAM	BLUEGILL	0.120	.	54.6
1975	277	CHEROKEE DAM	BLUEGILL	0.080	.	37.4
1975	277	CHEROKEE DAM	BLUEGILL	0.071	.	44.5
1975	277	CHEROKEE DAM	BLUEGILL	0.150	.	106.4
1975	277	CHEROKEE DAM	BLUEGILL	0.100	.	94.8
1975	277	CHEROKEE DAM	BLUEGILL	0.170	.	139.0
1975	277	CHEROKEE DAM	GIZZARD SHAD	0.063	.	149.1
1975	277	CHEROKEE DAM	GIZZARD SHAD	0.055	.	168.4
1975	277	CHEROKEE DAM	GIZZARD SHAD	0.027	.	230.5
1975	277	CHEROKEE DAM	GIZZARD SHAD	0.047	.	194.3
1975	277	CHEROKEE DAM	GIZZARD SHAD	0.056	.	213.3
1975	277	CHEROKEE DAM	GIZZARD SHAD	0.048	.	133.7
1975	277	CHEROKEE DAM	GIZZARD SHAD	0.069	.	136.4
1975	277	CHEROKEE DAM	GIZZARD SHAD	0.042	.	183.3
1975	277	CHEROKEE DAM	GIZZARD SHAD	0.056	.	250.7
1975	277	CHEROKEE DAM	GIZZARD SHAD	0.059	.	186.0
1975	277	CHEROKEE DAM	LARGEMOUTH BASS	0.058	.	23.9
1975	277	CHEROKEE DAM	LARGEMOUTH BASS	0.037	.	32.3
1975	277	CHEROKEE DAM	LARGEMOUTH BASS	0.054	.	45.9
1975	277	CHEROKEE DAM	LARGEMOUTH BASS	0.066	.	71.6
1975	277	CHEROKEE DAM	LARGEMOUTH BASS	0.049	.	74.9
1975	277	CHEROKEE DAM	LARGEMOUTH BASS	0.081	.	96.0
1975	277	CHEROKEE DAM	LARGEMOUTH BASS	0.070	.	162.8
1975	277	CHEROKEE DAM	LARGEMOUTH BASS	0.046	.	167.4
1975	277	CHEROKEE DAM	LARGEMOUTH BASS	0.150	.	191.2
1975	277	CHEROKEE DAM	LARGEMOUTH BASS	0.530	.	1020.5

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APPENDIX B. SUMMARY OF HOLSTON RIVER/CHEROKEE RESERVOIR WATER DATA

PART 1. SELECTED WATER CHEMISTRY DATA FOR STATIONS ON THE
HOLSTON RIVER AND CHEROKEE RESERVOIR FOR PERIOD MAY 1975
THROUGH JANUARY 1976. DOC=DISSOLVED ORGANIC CARBON (MG/L),
RIVER_KM = DISTANCE ABOVE (-) AND BELOW PLANT,
DIS_HG = DISSOLVED HG (MG/L), SS= SUSPENDED SOLIDS (MG/L),
FLOW_CFS = FLOW (CUBIC FT/SEC), REP_NO=REPLICATE SAMPLE,
CONDUCT = CONDUCTIVITY (UMHOS/CM),
CL = CHLORIDE (MG/L), TOT_HG = TOTAL HG (UG/L),
POC = PARTICULATE ORGANIC CARBON (MG/L).

DATE	RIVER_KM	STATION	TIME	FLOW_CFS	REP_NO	TOT_HG	DIS_HG	PH	CONDUCT	CL	DOC	POC	SS
9/10/75	TRIBUTARY	GERMAN CREEK	1745	.	0	0.009	.	8.4	253.00	14.6	2.70	2.99	41.6
9/10/75	TRIBUTARY	POOR VALLEY CR	1900	.	0	0.007	.	8.6	198.00	2.7	3.20	1.20	36.0
8/27/75	-43	CHATHAM HILL	.	.	0	0.070	0.001	8.0	248.00	1.8	2.16	.	12.0
8/27/75	-14	OLD BROAD FORD	.	.	0	0.025	0.000	8.3	253.00	4.2	1.62	.	11.2
8/27/75	-9	NIELS BRIDGE	.	.	0	0.034	.	8.1	251.00	4.2	2.32	.	7.8
7/31/75	-5.5	MC GRADY CREEK	.	1.0	0	0.023	.	.	.	2.3	.	.	.
7/29/75	-2.7	USGS SALTVILLE	1310	60.0	0	0.006	0.000	8.0	246.00	2.7	3.15	0.58	16.8
7/30/75	-2.7	USGS SALTVILLE	1300	57.0	0	0.007	0.000	8.2	244.00	2.5	2.05	0.32	12.0
7/31/75	-2.7	USGS SALTVILLE	1800	55.5	0	0.009	0.000	8.1	266.00	2.5	3.31	0.56	12.4
8/28/75	-2.7	USGS SALTVILLE	.	.	0	0.015	.	8.1	253.00	5.2	1.94	.	5.2
5/22/75	0	PERRYVILLE CRK	1000	.	0	.	.	7.8	3.96	165.0	.	.	.
7/31/75	0	SALTVA GRAY LN	.	.	0	0.028	.	.	.	178.0	.	.	.
7/31/75	0	PERRYVILLE CRK	.	5.0	0	0.200	.	.	.	20.0	.	.	.
5/21/75	3.1	SALTVILLE TR S	1720	.	1	0.064	0.034	8.6	925.00	235.0	.	.	4.1
5/21/75	3.1	SALTVILLE TR N	1720	.	2	0.082	0.043	8.7	350.00	140.0	.	.	9.5
5/21/75	3.1	SALTVILLE TR N	1720	.	3	0.240	0.104	8.8	975.00	252.0	.	.	8.7
7/30/75	3.1	SALTVILLE TR S	900	60.4	1	0.193	0.064	8.0	1300.00	375.0	2.46	0.22	6.6
7/30/75	3.1	SALTVILLE TR N	900	60.4	2	0.199	0.072	8.1	1310.00	394.0	1.97	0.37	10.0
7/30/75	3.1	SALTVILLE TR N	900	60.4	3	0.155	0.064	8.0	3220.00	1080.0	2.27	0.39	9.2
7/30/75	3.1	SALTVILLE TR	900	60.4	0	0.196	0.071	8.1	1405.00	427.0	2.01	0.36	9.8
8/27/75	3.1	SALTVILLE TR	.	.	0	0.210	0.056	8.2	5150.00	1490.0	2.71	.	11.2
7/29/75	6.4	RAILROAD TRESL	1530	67.6	0	0.167	0.075	9.3	4000.00	1275.0	2.98	0.92	12.8
7/30/75	6.4	RAILROAD TRESL	1235	64.2	0	0.185	0.050	8.0	1280.00	358.0	3.60	0.30	12.4
7/31/75	6.4	RAILROAD TRESL	1835	62.5	0	0.193	0.074	8.1	5700.00	1848.0	3.05	1.22	14.8
8/27/75	9.7	MCKENNA ISLAND	.	.	0	0.074	0.040	8.2	1179.00	300.0	2.36	.	4.2
8/27/75	17	SWINGING BRIDGE	.	.	0	0.058	0.028	8.2	1173.00	300.0	2.12	.	3.0
8/01/75	21	HAYTERS GAP	.	.	0	0.079	.	.	.	.	.	.	.
8/27/75	21	HAYTERS GAP	.	.	0	0.067	0.027	8.4	1250.00	334.0	3.83	.	6.2
8/01/75	37	HOLSTON US58BRG	.	.	0	0.036	.	.	.	.	.	.	.
8/27/75	43	HINES ISLAND	.	.	0	0.032	0.027	8.4	821.00	216.0	2.60	.	4.0
8/01/75	70	MENDOTA	.	.	0	0.038	.	.	.	.	.	.	.
8/01/75	119	GATE CITY	.	.	0	0.038	.	.	.	.	.	.	.
8/28/75	133	KINGSFORD	.	.	0	0.025	0.018	8.1	669.00	158.0	2.73	.	12.2
9/09/75	184	AMIS (BUREN BRG)	1615	.	0	0.010	.	7.6	354.00	43.2	3.28	1.59	4.8
9/09/75	191	SEVIER STEAM PL	1800	.	0	0.029	0.010	7.7	205.00	12.6	1.94	1.02	19.8
9/10/75	277	CHEROKEE-DAM SF	1100	.	0	0.006	0.005	8.5	224.00	14.1	1.81	0.70	3.8
9/10/75	277	CHEROKEE-DAM15M	1100	.	0	0.009	0.004	7.7	245.00	14.0	1.72	0.74	3.8
9/10/75	277	CHEROKEE-DAM28M	1100	.	0	0.012	0.006	7.4	267.00	15.0	1.86	0.93	21.0
9/10/75	278	CHEROKEE-TRAILWATER	1630	.	0	0.005	.	7.5	246.00	14.3	1.94	0.91	6.4
1/13/76	-2.7	USGS SALTVILLE	1500	212.0	1	0.012	0.000	7.8	160.00	.	.	.	1.5
1/13/76	-2.7	USGS SALTVILLE	1500	212.0	2	0.011	0.000	7.8	160.00	.	.	.	1.5
1/14/76	-2.7	USGS SALTVILLE	1115	644.0	1	0.009	0.000	7.7	145.00	.	.	.	62.0
1/14/76	-2.7	USGS SALTVILLE	1115	644.0	2	0.014	0.000	7.7	145.00	.	.	.	62.0
1/14/76	-2.7	USGS SALTVILLE	1815	674.0	1	0.008	0.000	7.6	142.00	.	.	.	37.2
1/14/76	-2.7	USGS SALTVILLE	1815	674.0	2	0.009	0.000	7.6	142.00	.	.	.	37.2
1/15/76	-2.7	USGS SALTVILLE	1800	512.0	1	0.004	0.000	7.5	132.00	.	.	.	11.2
1/15/76	-2.7	USGS SALTVILLE	1800	512.0	2	0.006	0.000	7.5	132.00	.	.	.	11.2
1/15/76	-2.7	USGS SALTVILLE	1800	512	3	0.004	0.000	7.5	132	.	.	.	11.2
1/15/76	-2.7	USGS SALTVILLE	1800	512	4	0.005	0.000	7.5	132	.	.	.	11.2
1/16/76	-2.7	USGS SALTVILLE	945	464	1	0.003	0.000	7.8	132	.	.	.	4.8
1/16/76	-2.7	USGS SALTVILLE	945	464	2	0.001	0.000	7.8	132	.	.	.	4.8
1/13/76	-2.7	USGS SALTVILLE	1500	212	0	0.012	0.000	7.8	160	.	.	.	1.5
1/14/76	-2.7	USGS SALTVILLE	1115	644	0	0.012	0.000	7.7	145	.	.	.	62.0
1/14/76	-2.7	USGS SALTVILLE	1815	674	0	0.008	0.000	7.6	142	.	.	.	37.2
1/15/76	-2.7	USGS SALTVILLE	1800	512	0	0.005	0.000	7.5	132	.	.	.	11.2
1/16/76	-2.7	USGS SALTVILLE	945	464	0	0.002	0.000	7.8	132	.	.	.	4.8
1/13/76	6.4	RAILROAD TRESL	1630	240	1	0.115	0.046	8.2	608	.	.	.	2.2
1/13/76	6.4	RAILROAD TRESL	1630	240	2	0.117	0.046	8.2	608	.	.	.	2.2
1/14/76	6.4	RAILROAD TRESL	1045	726	1	0.230	0.031	8.0	407	.	.	.	44.4
1/14/76	6.4	RAILROAD TRESL	1045	726	2	0.186	0.031	8.0	407	.	.	.	44.4
1/14/76	6.4	RAILROAD TRESL	1745	760	1	0.117	0.000	7.8	425	.	.	.	52.8
1/14/76	6.4	RAILROAD TRESL	1745	760	2	0.127	0.000	7.8	425	.	.	.	52.8
1/15/76	6.4	RAILROAD TRESL	1700	577	1	.	.	8.0	492	.	.	.	10.2
1/15/76	6.4	RAILROAD TRESL	1700	577	2	0.081	0.027	8.0	492	.	.	.	10.2
1/15/76	6.4	RAILROAD TRESL	1700	577	3	0.087	0.027	8.0	492	.	.	.	10.2
1/15/76	6.4	RAILROAD TRESL	1700	577	4	0.083	0.027	8.0	492	.	.	.	10.2
1/16/76	6.4	RAILROAD TRESL	1015	523	1	0.085	0.052	8.0	357	.	.	.	5.8
1/16/76	6.4	RAILROAD TRESL	1015	523	2	0.089	0.052	8.0	357	.	.	.	5.8
1/13/76	6.4	RAILROAD TRESL	1630	240	0	0.116	0.046	8.2	608	.	.	.	2.2
1/14/76	6.4	RAILROAD TRESL	1045	726	0	0.209	0.031	8.0	407	.	.	.	44.4
1/14/76	6.4	RAILROAD TRESL	1745	760	0	0.123	0.000	7.8	425	.	.	.	52.8
1/15/76	6.4	RAILROAD TRESL	1700	577	0	0.084	0.029	8.0	492	.	.	.	10.2
1/16/76	6.4	RAILROAD TRESL	1015	523	0	0.087	0.052	8.0	357	.	.	.	5.8

APPENDIX B. (continued)

PART 2. DATA FOR MERCURY IN THE SEDIMENTS OF THE HOLSTON AND CHEROKEE LAKE FOR SAMPLES COLLECTED BETWEEN FEBRUARY 1975 AND MAY 1977. SILTCLAY = % LESS THAN 44 MICROMETERS, PRAC_VOL = LOSS ON IGNITION (%), CS_137 = CESIUM 137 (DPM/G), TOT_HG = TOTAL HG (UG/L), RIVER_KM = DISTANCE ABOVE (-) AND BELOW PLANT.									
DATE	RIVER_KM	STA_NAME	SAM_TYPE	DEPTH_CM	SILTCLAY	PRAC_VOL	TOT_HG	CS_137	
2 75	TRIBUTARY	GERMAN CREEK	CORE SAMPLE	0-5	40	2.7	0.150	.	
2 75	TRIBUTARY	GERMAN CREEK	CORE SAMPLE	5-10	62	2.9	0.130	.	
2 75	TRIBUTARY	POOR VALLEY CR	CORE SAMPLE	0-5	11	2.5	0.080	.	
2 75	TRIBUTARY	POOR VALLEY CR	CORE SAMPLE	5-10	50	3.5	0.110	.	
2 75	216	GALBRAITH SPR	CORE SAMPLE	0-5	12	1.6	0.180	.	
2 75	216	GALBRAITH SPR	CORE SAMPLE	5-10	12	1.1	0.150	.	
5 75	-2.7	USGS SALTVILLE	CORE SAMPLE	0-3	.	.	0.110	.	
5 75	-2.7	USGS SALTVILLE	CORE SAMPLE	3-10	20	6.5	1.250	.	
5 77	277	CHEROKEE-DAM	CORE SAMPLE	0-5	.	.	0.570	3.10	
5 77	277	CHEROKEE-DAM	CORE SAMPLE	5-10	.	.	0.600	3.10	
5 77	277	CHEROKEE-DAM	CORE SAMPLE	10-15	.	.	0.690	4.40	
5 77	277	CHEROKEE-DAM	CORE SAMPLE	15-20	.	.	0.770	6.40	
5 77	277	CHEROKEE-DAM	CORE SAMPLE	20-25	.	.	0.830	7.50	
5 77	277	CHEROKEE-DAM	CORE SAMPLE	25-30	.	.	0.970	4.00	
5 77	277	CHEROKEE-DAM	CORE SAMPLE	30-35	.	.	1.130	2.00	
5 77	277	CHEROKEE-DAM	CORE SAMPLE	35-40	.	.	0.580	0.67	
5 77	277	CHEROKEE-DAM	CORE SAMPLE	40-45	.	.	0.220	.	
5 77	277	CHEROKEE-DAM	CORE SAMPLE	45-47	.	.	0.100	.	
8 75	-43	CHATHAM HILL	GRAB SAMPLE	0-2	.	9.0	0.130	.	
8 75	-43	CHATHAM HILL	GRAB SAMPLE	0-2	.	9.9	0.140	.	
8 75	-14	OLD BROAD FORD	GRAB SAMPLE	0-2	.	3.4	0.080	.	
8 75	-14	OLD BROAD FORD	GRAB SAMPLE	0-2	.	10.1	0.160	.	
8 75	-9	NIELS BRIDGE	GRAB SAMPLE	0-2	.	10.4	0.170	.	
8 75	-9	NIELS BRIDGE	GRAB SAMPLE	0-2	.	10.6	0.160	.	
8 75	3.1	SALTVILLE	GRAB SAMPLE	0-2	.	5.5	20.000	.	
8 75	3.1	SALTVILLE	GRAB SAMPLE	0-2	.	6.2	18.000	.	
8 75	3.1	SALTVILLE	GRAB SAMPLE	0-2	.	7.8	18.000	.	
8 75	9.7	MCKENNA ISLAND	GRAB SAMPLE	0-2	.	9.6	9.400	.	
8 75	9.7	MCKENNA ISLAND	GRAB SAMPLE	0-2	.	10.9	9.100	.	
8 75	17	SWINGING BRIDGE	GRAB SAMPLE	0-2	.	6.6	6.000	.	
8 75	17	SWINGING BRIDGE	GRAB SAMPLE	0-2	.	8.3	6.300	.	
8 75	21	HAYTERS GAP	GRAB SAMPLE	0-2	.	7.3	4.200	.	
8 75	21	HAYTERS GAP	GRAB SAMPLE	0-2	.	6.8	4.400	.	
8 75	43	HINES ISLAND	GRAB SAMPLE	0-2	.	6.5	3.000	.	
8 75	43	HINES ISLAND	GRAB SAMPLE	0-2	.	6.2	1.900	.	
8 75	133	KINGSPORT	GRAB SAMPLE	0-2	.	5.6	0.980	.	
8 75	133	KINGSPORT	GRAB SAMPLE	0-2	.	6.9	0.880	.	
9 75	TRIBUTARY	POOR VALLEY CR	GRAB SAMPLE	0-2	.	4.9	0.056	.	
9 75	TRIBUTARY	POOR VALLEY CR	GRAB SAMPLE	0-2	.	4.9	0.052	.	
9 75	TRIBUTARY	GERMAN CR	CORE SAMPLE	0-5	.	5.7	0.110	.	
9 75	TRIBUTARY	GERMAN CR	CORE SAMPLE	0-5	.	.	0.076	.	
9 75	194	AMIS (BOREM BRG)	EKMAN DREDGE	0-4	.	9.0	5.200	.	
9 75	198	SEVIER STEAM PL	EKMAN DREDGE	0-4	.	6.4	0.860	.	
9 75	198	SEVIER STEAM PL	EKMAN DREDGE	0-4	.	3.6	0.560	.	
9 75	198	SEVIER STEAM PL	CORE SAMPLE	0-5	.	5.3	1.000	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	0-5	23	.	0.840	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	5-10	24	.	1.900	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	10-15	85	.	8.100	1.36	
9 75	198	SEVIER LAKE	CORE SAMPLE	15-20	93	.	8.70	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	20-25	93	.	4.30	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	25-30	93	.	1.511	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	30-35	93	.	4.10	1.57	
9 75	198	SEVIER LAKE	CORE SAMPLE	35-40	67	.	4.60	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	40-45	66	.	4.60	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	45-50	90	.	4.00	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	50-55	91	.	3.20	1.27	
9 75	198	SEVIER LAKE	CORE SAMPLE	55-60	84	.	4.30	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	60-65	72	.	5.20	1.25	
9 75	198	SEVIER LAKE	CORE SAMPLE	65-70	69	.	3.80	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	70-75	67	.	4.20	1.24	
9 75	198	SEVIER LAKE	CORE SAMPLE	75-80	30	.	1.40	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	80-85	88	.	4.70	1.71	
9 75	198	SEVIER LAKE	CORE SAMPLE	85-90	83	.	4.90	.	
9 75	198	SEVIER LAKE	CORE SAMPLE	90-95	78	.	3.60	2.02	
9 75	198	SEVIER LAKE	CORE SAMPLE	95-100	41	.	2.70	.	
9 75	230	CHEROKEE-FALL CR	EKMAN DREDGE	0-4	.	10.2	1.70	.	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	0-5	.	9.4	2.10	.	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	0-5	.	.	2.40	1.36	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	5-10	.	.	3.20	1.41	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	10-15	.	.	4.10	1.69	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	15-20	.	.	5.80	1.50	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	20-25	.	.	8.10	1.60	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	25-30	.	.	3.60	2.08	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	30-35	.	.	4.20	1.96	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	35-40	.	.	3.70	2.54	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	40-45	.	.	3.80	3.07	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	45-50	.	.	4.00	2.82	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	50-55	.	.	3.90	2.88	
9 75	230	CHEROKEE-FALL CR	CORE SAMPLE	55-60	.	.	3.80	2.73	
9 75	238	CHEROKEE-25E BR	EKMAN DREDGE	0-4	.	.	1.60	.	
9 75	277	CHEROKEE-DAM	EKMAN DREDGE	0-4	.	.	0.47	.	
9 75	277	CHEROKEE-DAM	EKMAN DREDGE	0-4	.	.	0.48	.	

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